

LEACHING TESTS FOR ASSESSING MANAGEMENT OPTIONS FOR
INDUSTRIAL SOLID WASTE: A CASE STUDY USING ASH
FROM THE COMBUSTION OF WOOD AND TIRES

By

THABET MOHAMMED TOLAYMAT

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Thabet Mohammed Tolaymat

This document is dedicated to my parents Farouk and Shadia Tolaymat.

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Thabet Mohammed Tolaymat

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Research was performed to evaluate methodologies currently used to assess risk to human health and the environment through groundwater contamination from the leaching of inorganic contaminants from land-applied industrial waste. This was accomplished through the detailed characterization of one particular type of solid waste (wood and tire (WT) ash), followed by a critical evaluation of the results with respect to current methodologies for assessing risk from the leaching of contaminants to groundwater. The toxicity characteristic leaching procedure (TCLP), synthetic precipitation leaching procedure (SPLP) and deionized (DI) water extracted statistically similar concentrations of inorganic constituents. When compared to California's waste extraction test (WET), the TCLP extracted statistically lower concentrations of most inorganic constituents but statistically similar concentrations of barium, calcium and lead. Of the three factors that impact inorganic contaminant leachability examined, extraction pH was found to control metal leachability to the greatest extent.

Lysimeter leaching tests were conducted to better simulate actual environmental conditions. Statistically, the depth of the ash layer did not have an impact on the leachate quality generated by the lysimeters. Except for chromium, statistically similar concentrations of other metals were extracted by lysimeter and batch leaching tests. The lysimeter leachate extracted lower concentrations of ions at equivalent liquid-to-solid ratio than batch leaching tests, but based on the total mass extracted both methods extracted similar concentrations.

Comparing total metal content to theoretical back calculated risk-based standards for leaching found that arsenic, chromium and zinc posed an unacceptable groundwater contamination risk. The risk assessment using the SPLP, however, found aluminum, manganese, sulfate, total dissolved solids and pH to exceed their secondary drinking water standards (DWS). Lead exceeded its risk-based primary DWS in the SPLP leachate. Lysimeter results indicated that aluminum, pH, total dissolved solids (TDS), chlorides and sulfate exceeded their respective secondary DWS and lead exceeded its primary DWS.

The lysimeter leachate concentrations were used to examine a hypothetical land application scenario using the computer program MYGRT. The model indicated that under conditions of sufficient dilution, the land application of WT does not pose an unacceptable risk via groundwater. As a result with limitations with the existing approaches, an alternative approach for assessing risk from land-applied waste was presented. The approach relies on estimating the pore water concentration of the solid waste, and utilizes results from the SPLP and batch leaching test conducted at a much lower liquid-to-solid ratio to approximate the pore water concentrations.

CHAPTER 1 INTRODUCTION

1.1 Problem Statement

Environmental engineers, scientists and regulators utilize solid waste leaching tests to evaluate appropriate waste management options. For soil-like industrial solid wastes, the most often proposed management option outside landfill environments is land application. Since some of the constituents contained in a solid waste may be “bound” or otherwise immobile under typical disposal environments, the fraction of the constituents that “leach” (become mobile) from a solid waste is thought to be a better indicator of the potential risk to human health associated with certain solid waste management scenarios. In the context of environmental management, risk can be defined as the probability that public health may be adversely or unacceptably impacted from exposure to chemicals contained in the solid waste. One of the most common methods used to assess contaminant mobility from a solid waste is by conducting a leaching test.

The leaching of constituents from solid wastes has been studied for many years, and a number of laboratory-scale tests have been developed to predict contaminant leaching from waste in different management scenarios (Francis and Maskarinec 1986; Kosson et al. 2002; Van der Sloot et al. 1997). Simple leaching tests (batch leaching tests) rely on short-term leaching (less than 1 day) under static conditions and high liquid-to-solid ratios. More elaborate leaching tests (dynamic leaching tests) provide more realistic conditions of water flowing through waste and creating leachate; however these can be lengthy and costly.

The use of batch leaching tests for assessing the risk associated with a given waste management scenario can prove to be problematic, especially when a test is applied too broadly. Most of the short-comings associated with batch leaching tests stem from necessary assumptions regarding such factors as the tests' leaching contact time, liquid-to-solid ratio, and pH. Thus, a clearer understanding of these tests, as well as an evaluation of their use in assessing the risk associated with solid waste management, is needed.

1.2 Research Objectives

This research targeted the following objectives:

1. Characterize the inorganic chemical and mineralogic nature of ash from the combustion of wood and tires at a waste-to-energy facility in Florida and measure the wood and tire (WT) ash's leachable contaminant concentrations for a variety of inorganic constituents using standardized regulatory batch leaching tests.
2. Examine the impact of pH, leaching time, and liquid-to-solid ratio on the leachability of inorganic constituents from WT ash, and use this information to evaluate the appropriateness of using regulatory leaching tests to assess contaminant leachability under different waste management scenarios.
3. Simulate inorganic contaminant leachability that would occur from ash land application using dynamic leaching tests (lysimeters containing 1, 2, and 3 lifts of WT ash), and assess the effect of ash depth on leachate quality.
4. Compare the results of the static batch leaching tests and the dynamic lysimeter tests as a function of liquid-to-solid ratio.
5. Evaluate the risk of groundwater contamination from leaching of land-applied WT ash, A) using standardized regulatory practices and the results of the tests conducted, and then B) using available contaminant migration modeling software to model the migration of inorganic contaminants in groundwater.
6. Comment on the applicability of using batch tests for risk-based decision making, criticize the current regulatory risk evaluation technique.
7. Offer an alternative approach for assessing risk to groundwater contamination of land-applied solid waste.

1.3 Research Approach

In order to evaluate the leachability of industrial solid waste with respect to risks from beneficial use via land application, a non hazardous solid waste was selected. The chosen waste, wood and tire (WT) ash generated at a Florida waste to energy facility, met this requirement and also had a realistic potential for beneficial use in land application. While most organic contaminants in wood and tires are volatilized in the combustion process, inorganic contaminants (e.g., metals) become concentrated in the ash. This research concentrated on assessing the potential risk associated with inorganic contaminants present in the ash.

Representative ash samples were collected from a landfill where the WT ash was disposed. The ash was initially characterized by total metal analysis. The ash was then subjected to regulatory batch leaching tests that included the toxicity characteristic leaching procedure (TCLP), the synthetic precipitation leaching procedure (SPLP), waste extraction test (WET), multiple extraction procedure (MEP) and a deionized water extraction test. In addition, other batch leaching tests were designed to examine the effects of change in pH, contact time and liquid-to-solid ratio on metal leachability from the ash.

To better simulate actual field-leaching conditions, leaching column (lysimeter) tests were conducted. Ten stainless steel lysimeters were utilized: three lysimeters were filled with one foot of WT ash, three lysimeters were filled with two feet of WT ash, three lysimeters were filled with three feet of WT ash, and the final lysimeter was filled with sand empty as a control. Simulated rainwater was percolated through the lysimeters and leachate was collected and analyzed regularly. The effect of the depth of WT ash on leachate quality was examined. A detailed comparison between lysimeter leaching and

batch leaching tests was then conducted. As part of the risk assessment process, lysimeter leachate metal concentrations were used as an input into a contaminant transport model, MYGRT. The risk associated with the land application of wood and tire ash was first examined following procedures typically used by regulators. Contaminant concentrations calculated by the model were compared to those estimated by the current regulatory risk assessment procedure. Based upon the evaluation of that comparison, an alternate tiered approach to assess risk to groundwater by land applying solid waste was introduced.

1.4 Dissertation Organization

This dissertation is organized into 9 chapters. Chapter 1 provides the justification for the research along with the objectives. Chapter 2 reviews the literature pertaining to the different leaching tests used for assessing risk to groundwater contamination; discusses factors impacting inorganics leachability; outlines the current risk assessment process used by regulators; and examines the historical literature characterizing wood ash and tire ash. Chapter 3 presents the materials and methods used in this work and includes a brief discussion of the batch leaching tests, a description of the lysimeter leaching tests, the statistical analysis used throughout the research and a discussion of the groundwater flow models used.

In Chapter 4, the inorganic chemical and mineralogical nature of the WT ash and the results of the regulatory leaching tests are presented, as well as an assessment of inorganic contaminant leachability factors (pH, contact time and liquid-to-solid ratio). Chapter 5 presents the results of the lysimeter leaching study and an analysis of the effect of the depth of ash on leachate quality. Chapter 6 is a comparison of the results of the batch and lysimeter leaching tests. Chapter 7 includes the risk assessment process

currently used by regulators and a presentation of the groundwater modeling results.

Chapter 8 gives a detailed outline of a proposed screening method for assessing risk of groundwater contamination. Chapter 9 summarizes the major findings and conclusions of this research.

CHAPTER 2 BACKGROUND INFORMATION

2.1 Strategies and Requirements for Solid Waste Management

Solid waste encompasses a wide variety of materials generated by human activities, including household and commercial refuse (municipal solid waste (MSW)), chemical waste from industrial activities, waste produced during the treatment of polluted water or air, and waste from agriculture and mining (Environmental Protection Agency [EPA] 2002a). The majority of solid waste generated in the United States is managed through landfilling and incineration. In an effort to protect human health and the environment from the potential hazards of industrial waste disposal, Congress passed the Resource Conservation and Recovery Act (RCRA) in 1976. The Act defined wastes that were "hazardous" to human health and the environment so that appropriate regulations could be enacted to control the hazards these wastes presented.

2.1.1 Management of Hazardous Waste

A solid waste is a hazardous waste by U.S. federal regulation under a number of circumstances (Code of Federal Regulations [CFR] 2003b). If a solid waste is included on any of a number of specific lists, the waste is hazardous. To prevent circumvention of proper management requirements of listed hazardous wastes, these wastes remain hazardous regardless of the concentration of "hazardous" constituents (thus dilution is not a permitted management option). Since not every conceivable waste representing a hazard to human health and the environment can be listed, a set of hazardous characteristics is also defined. If a waste meets one or more of four characteristics

(ignitability, corrosivity, reactivity, and toxicity), the waste is characterized as hazardous. Each characteristic is defined and can be specifically tested for according to the regulations. For the toxicity characteristic, a single extraction test was not readily available and agreed upon, so a leaching test (the toxicity characteristic leaching procedure or TCLP) was developed. The TCLP simulates a scenario in which an industrial waste is co-disposed with organic wastes. If the concentration of a leached chemical exceeds a regulatory limit, the waste is considered hazardous by the toxicity characteristic (TC).

The US Environmental Protection Agency EPA mandates that hazardous waste be managed by an extensive set of record keeping and recording requirements, as well as a number of engineering controls (e.g., requirement of double lined landfills for land disposal of hazardous wastes) (CFR 2003c). As the U.S. solid and hazardous waste management system has evolved, the potential for other types of solid wastes (those not listed or characterized as hazardous) to impact the environment in certain situations has begun to be recognized.

2.1.2 Management of Municipal Solid Waste (MSW)

Americans disposed of an estimated 232 million tons of MSW, also commonly known as trash or garbage, in the United States in 2000. Approximately 55.3% of the garbage was disposed of in landfills and 14.5% was combusted, while the rest (30.2%) was recycled and composted (EPA 2002a). The US EPA promulgated regulations for MSW landfills in the early 1990's, imposing location restrictions and requiring that these facilities to possess liner systems, groundwater monitoring, and financial assurance. More recently, rules were promulgated for other types of wastes (e.g., construction waste,

industrial waste) that contain small amounts of hazardous constituents. These rules establish requirements such as site control and groundwater monitoring.

Currently all MSW landfills are regulated under the RCRA. RCRA established various design and operating standards, groundwater monitoring, corrective action measures, and conditions (including financial requirements) for closing municipal landfills and providing post-closure care. State programs regulating landfills are required, however, to incorporate the federal regulations into the state regulations. While the federal regulations establish a framework and national minimum standards for protecting human health and the environment, the implementation of solid waste regulations remains the responsibility of state and local governments.

2.1.3 Management of Industrial Non-Hazardous Waste

The US EPA defines industrial waste to consist of a wide variety of non-hazardous materials that result from the production of goods and products (EPA 2002c). Each year, approximately 7.6 billion tons of non-hazardous industrial solid waste are generated and disposed of at a broad spectrum of industrial facilities (EPA 2002a). States, and some local, governments have regulatory responsibility for ensuring proper management of these wastes, and the management programs can vary considerably. Much of this industrial waste is disposed of in landfills, waste piles, and surface impoundments. However, because of the long-term care and high cost associated with disposal, there is a trend in the industry to beneficially use this type of waste. One of the major beneficial use options for soil-like industrial waste (e.g., ashes, drinking water sludge and foundry sand) is land application. This includes use as road embankment, road base, fill material and soil amendment. In 2002, the US EPA realized the potential impact of land application of industrial waste on the

environment and issued a guidance for the management of non-hazardous industrial wastes (EPA 2002c).

2.2 Beneficial Use of Solid Waste Through Land Application

The beneficial use of non-hazardous industrial wastes promotes resource conservation and preserves landfill space. The beneficial use typically proposed for these wastes is land application, including use as a soil amendment, fill material and road embankment material (Campbell et al. 1990). Examples of wastes often used beneficially include combustion ash, fines from the recycling of construction and demolition (C&D) debris, foundry sand, scrap tires, and reclaimed concrete (Lerner and Utzinger 1986; Naylor and Schmidt 1986; Schreurs et al. 2000; Townsend 1998; Vance 1996). As discussed previously, current US federal regulations developed under the authority of RCRA detail the requirements for the management of hazardous industrial wastes. While guidance has been issued for the management of non-hazardous industrial wastes (EPA 2002c), no encompassing federal regulation exists for the beneficial reuse via land application for industrial wastes. However, rules have been promulgated for several specific wastes, including municipal biosolids (CFR 2003d) and cement kiln dust (Federal Register [FR] 2001). Since non-hazardous wastes may still contain pollutants at concentrations that may cause unacceptable risk to human health and the environment, some strategy for determining appropriateness of beneficial use should be followed.

With the lack of federal guidance in this regard, some states have developed guidelines for the beneficial use of industrial wastes through land application (Florida Department of Environmental Protection [DEP] 1998; DEP 2001; New York Code of Regulations [NYCR] 2002; Wisconsin Administrative Code [WAC] 2002). For example,

wood ash may be used as a fertilizer and a soil amendment (after proper testing) in New York State (NYCR 2002). Wisconsin and Iowa allow coal bottom ash, coal fly ash and paper mill residue to be used as a soil amendment or as a fill material if contaminant concentrations are below use guidelines (Iowa Administrative Code [IAC] 2003; WAC 2002). Florida developed guidelines for the beneficial use of fines generated from C&D debris recycling (DEP 1998) and waste to energy ash (DEP 2001). Iowa allows the use of foundry sand in road construction (IAC 2003). The guidelines of these states typically rely on some form of risk assessment in order to determine whether a specific beneficial use scenario is environmentally acceptable.

One method to evaluate the risk posed from beneficial use of solid wastes is a site-specific risk assessment. Site-specific risk assessments, however, involve detailed waste characterization, contaminant migration modeling, exposure routes and consider site characteristics such as hydrology. Consequently, a site-specific risk assessment can be costly and time intensive.

Generic contaminant concentrations (often referred to as screening levels, guidance concentrations or target concentrations) have been established by many states to provide a quick and relatively inexpensive method to evaluate risk. Above these concentrations, beneficial use is often restricted, as contaminants may pose an unacceptable risk to human health and the environment. The methodology behind these limits was originally developed to assess and categorize the cleanup of contaminated sites, but it is now widely employed for setting limits for wastes beneficially used in a similar manner as soil. These limits vary from state to state, reflecting different background contaminant concentrations, exposure assumptions and acceptable risk levels. Nonetheless, the

majority of these guidelines are based on a similar methodology as outlined in the Environmental Protection Agency's soil screening guidance (SSG) (EPA 1996a), which was established for the remediation of Superfund sites. The SSG includes a set of conservative limits (soil screening levels (SSL)) for risk assessment (EPA 2001).

The SSG document and most of the states' guidelines assess the potential for human impact from contaminants through two common exposure routes: direct human exposure and exposure through the consumption of contaminated groundwater. Direct human exposure (inhalation, dermal contact and ingestion) relies on the conservative assumption that all, or a defined amount, of the contaminant present in a waste is available for uptake. Thus the total concentration of a waste pollutant (mg/kg) is compared directly to these risk-based limits for a given exposure scenario (such as a residential or an industrial setting). Residential screening limits evaluate exposure in a residential area where, for example, a child may ingest the contaminated media. The industrial setting evaluates exposure to workers during an 8 hour workday.

Evaluating human exposure through the consumption of contaminated groundwater is somewhat more complicated. This scenario has to take into consideration not only the amount of pollutants leaching from a solid waste but also the interactions with surrounding soil and the groundwater. Exposure through this route is generally evaluated either by comparing total contaminant concentrations (mg/kg) to a theoretical leaching based screening limit, or by comparing leachable concentrations (mg/L) from a leaching test to risk-based groundwater limits. Since it is difficult to account for all possible interactions with the surroundings, there are still many unknowns in assessing the risk to groundwater contamination from the land application of solid wastes. A simplistic

schematic of the risk assessment process followed by most regulators is presented in Figure 2-1.

2.3 Solid Waste Management Leaching Tests

Important tools often used in the decision making process of solid waste management are leaching tests. Leaching tests are used to estimate the potential concentration of contaminant that leaches from a solid waste when exposed to water or similar medium. In general, there are two types of leaching tests performed: static (batch) and dynamic (lysimeter) leaching tests. While batch leaching tests are characterized by a constant liquid-to-solid ratio, dynamic (column or lysimeter) leaching tests are characterized by increasing liquid-to-solid ratio as the experiment proceeds. Batch leaching tests involve the mechanical mixing of a given volume of extraction solution with a unit mass of solid waste. Although batch tests offer a relatively rapid, cost-effective method for the assessment of metal leachability, the wide and sometimes inappropriate application of their results have lead to the increasing criticism of their use (Kosson et al. 2002).

The manner in which dynamic leaching test are conducted offer a more accurate simulation of actual field conditions when compared to batch leaching tests (Kosson et al. 2002; Van der Sloot et al. 1997). The continuous addition of extraction solution to a constant mass of solids in the lysimeter allows for an increasing liquid-to-solid ratio that is more representative of field conditions. Also, when compared to batch leaching tests, the slow movement of the extraction fluid through the solid waste in the lysimeter leaching tests more closely represents field conditions. As a result, lysimeter leaching tests are often used to verify batch leaching tests results and to examine contaminant concentrations in the field (Francis and Maskarinec 1986; Francis et al. 1984). To validate the development of the TCLP, Francis et al. (1984) compared contaminant

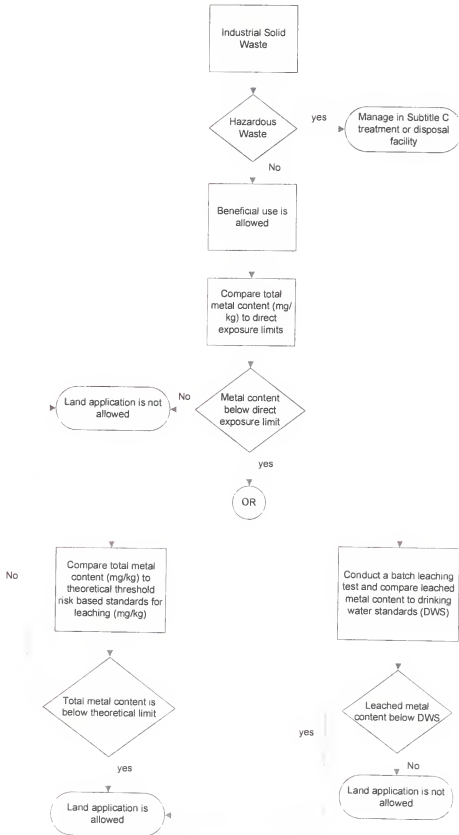


Figure 2-1: Schematic of Risk Assessment Process for Land Application of Solid Wastes

concentrations in lysimeter leachate, in which an industrial waste was co-disposed with MSW, to concentrations extracted by the TCLP.

2.3.1 Factors Affecting Metal Leachability

While batch leaching tests predict metal leachability under a given disposal or reuse scenario, they are not specific to predicting long-term metal leachability. Guidelines for utilizing and interpreting these or other tests are needed to determine the environmental effects on metal migration from a land-applied waste such as ash. Unlike prescribed regulatory tests, long-term and other leaching tests have the advantage of changing variables, such as the pH, liquid-to-solid ratio and the contact time. The regulatory test methods discussed previously require an initial extraction solution pH. Thus, once the extraction begins, pH may change, depending on the initial pH and alkalinity of the waste.

Equilibrium pH may be the single most important factor in determining metal leachability and migration (Buchholz and Landsberger 1995b; Cernuschi et al. 1990; Herbert et al. 1995; Kanungo and Mohapatra 2000; Kosson et al. 2002; Van der Sloot et al. 2001). Studies have demonstrated that as pH decreases, metal desorption increases; thus, metals are more soluble under acidic conditions. When the extraction pH moves to neutral - alkaline range, metal leachability decreases, as the metal ions tend to sorb back onto the solid phase. In a study examining metal leachability of coal fly ash, lead leached 2 mg/kg at a pH of 3, while it leached at a concentration of 0.6 mg/kg when the pH was raised to 8 (Herbert et al. 1995). Van der Sloot (2002) observed that the zinc concentration leached from cement stabilized fly ash dropped from 200 mg/L to less than 0.01 mg/L when the pH changed from 4 to 9. Under extreme alkaline conditions,

however, the leachability of some metals increases again. A study conducted on ash, from an iron recycling industry, showed the zinc concentration to decrease from approximately 8 to 0.001 mg/L as the pH changed from 5.5 to 7.5, and then increase back to approximately 0.01 mg/L as pH increase to 8 (Kjledsen and Christensen 1990).

Similar behavior was observed in lead and zinc release from MSW bottom ash (Van der Sloot et al. 1997). The increase in metal concentrations may be attributed to the tendency of these metals to form soluble metal hydroxide oxides in these environments (Fytianos et al. 1998; Youcai et al. 2002).

Along with pH, liquid-to-solid ratio also plays an important role in controlling metal leachability. For a fixed metal concentration in a solution, adsorption sites increase as the solid phase concentration increases. Thus, metals have the tendency to adsorb back onto the solid phase (Calvet et al., 1990; Bordas et al., 2000). When liquid-to-solid ratio decreased from 10,000 to 20 L/kg, researchers observed a 20 to 50% non-linear increase in cadmium and zinc leachability (Bordas and Bourg 2001). In another study conducted on coal fly ash, a change in liquid-to-solid ratio from 5 to 15 was associated with a 1% drop in lead concentration (Fytianos et al. 1998). Buchholz and Landsberger (1995b) observed that a changing in liquid-to-solid ratio from 2 to 6 was accompanied by a drop in the barium concentration, leached from MSW fly ash, from 10 to less than 1 mg/L. Understanding the change in metal release as a function of liquid-to-solid ratio is important especially when comparing two leaching tests like batch and lysimeter leaching tests.

Another factor controlling metal leachability from solid wastes is contact time. A long contact time allows for the more slow oxidation reduction reaction to occur. In a

study examining the effect of contact time on metal leachability from a contaminated soil, the batch leaching test oxidation reduction potential decreased from 250 to -150 mv as the contact time increased from 1 hr to 500 hours. As a result of the change in oxidation reduction potential the authors observed an increase in lead release from 0.3 to 0.7 mg/kg (Chuan et al. 1996). The leached iron concentration increased by 3 folds when the batch leaching tests contact time increased from 1 to 20 days (Kanungo and Mohapatra 2000). Because of the long contact time associated with lysimeter leaching test (often weeks) relative to batch leaching tests (often less than one day), understanding of the effect of contact time on metal leachability is also important in the comparison of these two tests types to each other.

2.3.2 Batch Leaching Tests

The potential for contaminant release is frequently estimated by conducting batch leaching tests. In general, these tests encompass a specific amount of solid waste and extraction fluid in controlled contact for a prescribed time frame. Batch tests are primarily conducted as a regulatory tool (e.g., to determine whether a solid waste is hazardous or not), but they can also be conducted in a manner to examine a specific variable's influence on metal mobility.

2.3.2.1 Toxicity characteristic leaching procedure (TCLP)

Of the regulatory prescribed batch tests, the TCLP is used most frequently. It is the leaching procedure required for characterizing waste as hazardous for the toxicity characteristic (TC) under RCRA (CFR 2003a; Francis and Maskarinec 1986; Francis et al. 1984). The TCLP evaluates metal leachability in municipal solid waste landfills. The extraction fluid simulates a scenario in which an industrial waste is co-disposed with

MSW (Kosson et al. 2002). The primary extraction fluid is a buffered acetic acid solution in a liquid-to-solid ratio of 20:1 rotated for eighteen hours. The concentrations of contaminants (organic and 8 inorganic) are compared to TC limit concentrations. The TC limit for the 8 inorganic contaminants is presented in Table 2-1.

Table 2-1: RCRA Toxicity Characteristic Limits

Metal	RCRA TC Limit (mg/L)
As	5.0
Ba	100
Cd	1.0
Cr	5.0
Hg	0.2
Se	1.0
Ag	5.0
Pb	5.0

Because of the chelating ability of the acetic acid used in the TCLP, it often extracts more metals than deionized water alone (at similar pHs). The TCLP extracted a maximum of 17%, 47% and 58% of total cadmium, lead, and zinc while deionized water extracted a maximum of 12%, 0.3% and 3%, respectively, from wood ash (Sawhney and Frink 1991). In another study conducted on electroplating sludge, the TCLP extracted 372 and 28 mg/L of zinc and copper, respectively, while deionized water only extracted 1.8 and 0.87 mg/L of these metals respectively (Chang et al. 2001). When TCLP is compared to MSW leachate, Cernuschui (1990) found that TCLP extracted only 1.4 mg/L while MSW leachate extracted 50.2 mg/L of lead. However, Hooper et al. (1998) showed that the TCLP extracted metals such as lead, cadmium and zinc at greater levels than MSW leachate. A detailed analysis of factors that effect metal leachability may help in understanding this contradiction.

2.3.2.2 Waste extraction procedure (WET)

Another batch test that is used for hazardous waste characterization is California's waste extraction test (WET). Unlike the TCLP, the WET is conducted using a 10:1 liquid-to-solid ratio and the slurry is rotated for 48 hours. However, similar to the TCLP, the WET utilizes a buffered organic acid solution (citric acid). Hopper et al. (1998) compared the WET and the TCLP to MSW leachate and concluded that the WET simulates MSW leachate metal extractability better than TCLP especially with certain oxyanions such as arsenic and chromium. This may have been caused by the greater chelating ability of citric acid, used in WET, when compared to acetic acid, used in TCLP.

2.3.2.3 Synthetic precipitation leaching procedure (SPLP)

Both the TCLP and the WET simulate leaching that might occur in a co-disposal scenario where ash is co-disposed with MSW. For the case of land application, those two tests would overestimate the leachability of heavy metals. Another batch test, the synthetic precipitation leaching procedure (SPLP), simulates rainwater instead of leachate (FR 1998). The SPLP is used to evaluate the potential for metal leaching into ground and surface waters (Brantley and Townsend 1999; Hageman et al. 2000; Jang and Townsend 2001; Townsend et al. 2002). The time and liquid-to-solid ratio are the same as TCLP, but this batch method provides a more realistic assessment of metal leachability caused by rainfall, as the extraction fluid is intended to simulate precipitation or acid rain.

2.3.2.4 Other leaching procedures

The MEP simulates leaching caused by the repetitive precipitation of acid rain water (EPA 1996b). Although MEP may be used to address long-term leachability, it

was designed for the delisting of hazardous solid wastes (EPA 2001). The method mainly involves extracting a solid waste for a duration of seven days. This test combines aspects from both the TCLP and the SPLP. Extraction on the first day calls for TCLP extraction fluid at a liquid-to-solid ratio of 20:1; thereafter, the procedure calls for SPLP extraction solution at the same liquid-to-solid ratio. Because of the variety of factors, presented earlier, that effect metal release in batch leaching tests, the US EPA Science Advisory Board (SAB) concluded that to improve the interpretation and application of batch leaching tests, they should be verified with environmental data or other tests that more closely simulate, like lysimeter tests, actual environmental conditions (EPA 1991; EPA 1999b).

2.3.3 Dynamic Methods for Assessing Metal Leachability

Unlike batch leaching tests, which are performed under controlled environments (e.g., liquid-to-solid ratios and contact time), dynamic leaching tests, such as lysimeter tests, offer a more accurate simulation of actual field conditions (Kosson et al. 2002; Van der Sloot et al. 1997). Lysimeter leaching tests are characterized by an increasing liquid-to-solid ratio as the experiment proceeds, which results from the continuous addition of leaching fluid to a constant mass of solid waste. Compared to mechanical mixing (batch tests), the process of a liquid moving slowly through a static waste (lysimeter tests) at dynamic liquid-to-solid ratios more closely simulates environmental conditions. Thus, lysimeter leaching tests are often used to verify batch leaching tests and to predict contaminant fate in the environment (Francis and Maskarinec 1986; Francis et al. 1984; Jackson and Bisson 1990; Jackson et al. 1984). The main disadvantages of lysimeter studies are the associated costs and the long duration. Unlike most batch leaching tests,

which are usually carried out within days, lysimeter studies are sometimes conducted over several years. The long duration of lysimeter tests, however, allows for interactions of the waste with atmospheric carbon dioxide and provides adequate time for the more gradual oxidation-reduction reactions to occur (Jackson et al. 1984; Van der Sloot et al. 1997).

The validity of comparing lysimeter test results to batch leaching test results depends largely on the waste characteristics and the metals of concern. Since lysimeter tests are open, atmospheric interactions must be addressed. For example, the inflow of carbon dioxide into a waste during lysimeter leaching tests may affect the extraction pH of the waste, especially a waste with a low buffering capacity. This atmospheric interaction would cause a variation between the lysimeter and batch leaching tests results for the same waste (Fallman 2000). The effect of carbon dioxide on extraction pH, however, is not as evident if the examined waste is relatively alkaline and buffered (Cernuschi et al. 1990; Hjelm 1990; Jackson and Bisson 1990). Since various metals behave differently, the metals of interest in the waste examined are also a factor affecting the comparison of batch leaching tests with lysimeter tests.

The oxidized environment of most batch leaching tests may cause difficulty in the interpretation of leaching results for certain metals such as chromium and arsenic. In an examination of arsenic leachability from sewage sludge, Meng et al. (2001) concluded that, during batch leaching tests, the presence of oxygen oxidized the iron which, in turn, released the bound arsenic. Research conducted on electroplating sludge observed that batch tests overestimate chromium mobility when compared to a lysimeter study (Jackson et al. 1984). It was concluded that because of the oxidizing conditions in the

batch leaching test, chromium was in the more mobile hexavalent form while in the lysimeters it was in its reduced and relatively immobile trivalent form.

2.4 Leaching Tests for Assessing the Potential for Groundwater Contamination

Most states have developed risk-based groundwater limits (GWL). Often times several different contaminant limits are defined to reflect different possible groundwater uses. The most stringent scenario is where the water is assumed to be used for direct human consumption. Often in this case, the US EPA primary and secondary drinking water standards (Table 2-2) are adopted by these states as the groundwater quality limits. The states may add additional contaminants to this list with concentration limits based on established toxicological information. Since metals present in a land-applied solid waste may leach from the waste and migrate to the groundwater (Kosson et al. 2002; Man-Chi Lo et al. 2000; Steenari et al. 1998; Townsend et al. 2002), possible risk to groundwater

Table 2-2: US EPA Drinking Water Standards for Metals

Primary		Secondary	
Contaminant	Standard (µg/L)	Contaminant	Standard (µg/L)
As	0.05 (0.01 ^a)	Al	0.2
Ba	2	Cu	1.0
Be	0.004	Fe	0.3
Cd	0.005	Mn	0.05
Cr	0.1	Zn	5
Cu	1.3	Cl	250
Pb	0.015	SO ₄	250
Sb	0.006	TDS	500
Se	0.05	pH	6.5-8.5
Tl	0.002		

^astandards goes into effect in 2006

from land application of solid wastes should be assessed. The risk to groundwater contamination, however, is hard to approximate unless the contaminant concentration in pore water volume as it leaves the land-applied waste is estimated.

2.4.1 Pore Water and Compliance Point Concentrations

The risk to groundwater from a land-applied solid waste can be assessed by comparing the metal concentrations expected to occur in the groundwater (as a result of waste leaching and subsequent pollutant migration) to risk-based limits. However, several questions must be answered. First, at what point in the groundwater must the water quality meet the appropriate limits? This could be directly under the waste or it could be some compliance point away from the waste. Second, how does one predict the concentration expected to occur at the compliance point as a result of the waste? To better explain these points, a hypothetical land application scenario is presented in Figure 2-2 where a solid waste is land-applied as fill material.

Contaminant concentrations at three locations (A, B and C) are of particular interest. Contaminant concentrations in the leachate at the bottom of the land-applied waste, point A in Figure 2-2 (C_L), could be (but are not necessarily), the concentrations at the compliance point (C_{RC}). The compliance point could also be in the groundwater directly beneath the application boundary (point B), although it is often down gradient (point C). Several different methods will be discussed that may be used to estimate concentrations at point A (C_L); however, the concentrations expected to occur at points B or C are difficult to predict. As metals migrate away from the waste (point A), they may become attenuated through mechanisms such as sorption onto soil particles and diluted by mixing with uncontaminated groundwater. These mechanisms tend to reduce contaminant concentrations at points B and C relative to those at point A. This factor, also referred to as the dilution attenuation factor (DAF), is defined as the ratio of C_L to C_{RC} as presented in equation 2-1.

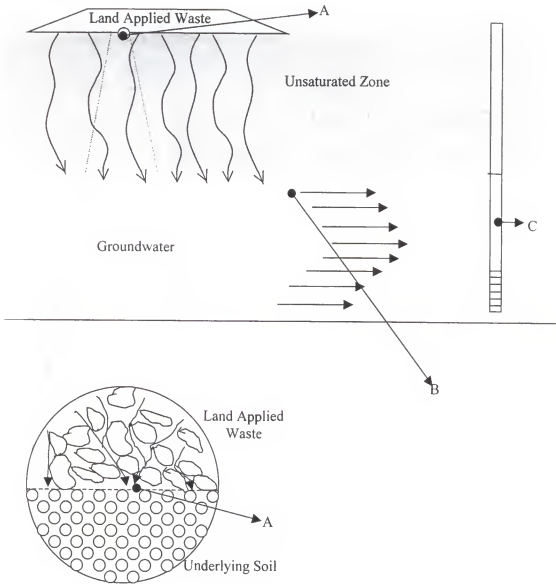


Figure 2-2: Schematic of Land Application Scenario

$$DAF = \frac{C_L \left(\frac{\text{mg}}{\text{L}} \right)}{C_{RC} \left(\frac{\text{mg}}{\text{L}} \right)} \quad (2-1)$$

A DAF can be obtained on a site-specific basis by collecting soil and hydrologic information, and studying the specific interactions between the waste leachate in question

and the soil. A more common approach, however, is to apply a generic DAF. The risk-based groundwater limits in many states utilize a DAF developed by the EPA for the Superfund soil screening guidance (SSG) (EPA 2001). In its development of the SSG, the EPA used the Composite Model for Leachate Migration with Transformation Products (EPACMTP) computer program. The model was applied on 300 sites in throughout the US to determine values for DAF as a function of the area of the contaminated site at various probability levels. The model examined groundwater concentrations at a compliance point at the site boundary (point C) and not underneath the contaminated soils (points A or B).

Although attenuation may lead to lower metals concentrations at point C (relative to points A or B), attenuation was not accounted for in the model and thus it reflects only groundwater dilution effects (EPA 2001). As a result of the modeling efforts, the EPA recommended a DAF of 20 (the average of sites that with an area of 0.5 and 30 acres) to be used as a conservative estimate (EPA 2001). It is noteworthy that earlier drafts of the SSG suggested using a DAF of 10 (EPA 1996a), and this value is still used by some states, such as Massachusetts and Iowa (CMR 2002; IAC 2003). In general, a typical approach to assess risk associated with the land application of a solid waste is to utilize a generic DAF to estimate C_{RC} based on C_L . Several methods have been used to estimate C_L , including a partitioning coefficient approach, batch leaching tests, and column leaching studies, which are discussed in the following section.

2.4.2 Partitioning Coefficient Approach

One method to predict C_L (mg/L) involves using the total amount of pollutant (mg/kg) present in the waste and an assumed partitioning coefficient (k_d). The partitioning coefficient is the ratio of contaminant in the solid phase to that in the liquid

phase (L/kg) (Dimitrova and Mehandgiev 1998; EPA 1996a; Thibault et al. 1990). To calculate total contaminant concentrations, the applicable groundwater limits are first multiplied by a DAF to obtain a target C_L concentration. For example, if the dilution factor is 20 and the acceptable groundwater concentration is 0.05 mg/L, the target C_L would be 1 mg/L. A simple equation is then used to calculate the total contaminant concentration (i.e., SSL for leaching) in the waste corresponding to this pore water concentration as presented in Equation 2-2.

$$SSL_{Leaching} = GWL \times DAF \times k_d \quad (2-2)$$

where $SSL_{Leaching}$ is the soil screening level for leaching, GWL is the groundwater limit for the particular contaminant of interest, DAF is the dilution attenuation factor, and k_d is the partitioning coefficient. The US EPA's generic $SSL_{Leaching}$ for inorganic contaminants are presented in Table 2-3.

This allows for affordable testing since total metal concentrations, which are relatively quicker to conduct than batch leaching tests, are directly compared to the $SSL_{Leaching}$ presented in the SSG and various state guidelines. The SSLs for leaching used by most states for risk-based waste management decision-making are based on values computed during the development of SSG. This approach, however, is largely dependent on the estimation of k_d values. The k_d values for metals are significantly affected by a variety of environmental conditions. The most significant parameters, as discussed earlier in section 2.3.1, are pH (Herbert et al. 1995) and oxidation-reduction conditions (Chuan et al. 1996). The number of significant influencing parameters, their variability in the field, and differences in experimental methods result in a wide range of

k_d values for individual metals reported in the literature (Baes and Sharp 1983; Baes et al. 1984; Thibault et al. 1990).

Table 2-3: US EPA's Generic SSL for Leaching

Compound	SSL _{Leaching} (DAF) = 20 (mg/kg)	SSL _{Leaching} (DAF) = 1 (mg/kg)
Antimony	5	0.3
As	29	1
Ba	1,600	82
Cd	8	0.4
Cr	38	
Pb	NA	NA
Hg	2	0.1
Ni	130	7
Se	5	0.3
Ag	34	2
Tl	0.7	0.07
V	6,000	300
Zn	12,000	620

While an equilibrium geochemical speciation model (MINTEQ2) was used to generate k_d values for most metals, analytical pH-dependent adsorption relationships were used for arsenic, hexavalent chromium, selenium and thallium. The k_d values at a pH of 6.8 were used as default values in the development of the SSLs, and a few are presented in Table 2-4. While this method of assessing risk to groundwater contamination is rather cheap and quick, it relies heavily on k_d values developed for soils at a pH of 6.8. The pH and k_d values might be different for various solid wastes. Because of the dependence of this approach on the estimated k_d value, other approaches

Table 2-4: Partitioning Coefficient for Selected Metals at pH of 6.8

Metal	k_d at pH = 6.8 (L/kg)
As	29
Ba	41
Be	790
Cd	75
Cr	19
Zn	62

(e.g., batch leaching tests) that examine release from the actual waste tested are often used.

2.4.3 Batch Leaching Tests Approach

Another approach to examine metal leachability from solid wastes is to conduct batch leaching tests. However, because of the relatively high liquid-to-solid ratio associated with batch tests, some state guidelines (e.g., Florida, Massachusetts) assume that some dilution occurs in the leaching test. These guidelines assume that the SPLP leachate contaminant concentrations represent actual groundwater concentrations (C_{RC}). Consequently, it is a common practice in these states to compare the results of a batch leaching test such as the SPLP directly to groundwater standards to determine whether a contaminated soil or land-applied waste will present a risk through groundwater contamination (Saranko et al. 1999).

Contrary to this, the EPA assumes that the SPLP leachate contaminant concentrations are those encountered directly below the waste C_L (Figure 2-2 point A). As a result, the EPA recommends applying a DAF of 20:1 to SPLP leachate contaminant concentration to estimate C_{RC} (points B and C) (EPA 2001). The DAF factor used here was adopted from the same method presented earlier in section 2.4.1. Although batch leaching tests utilize the actual land-applied waste, the degree of dilution is still unclear. It is still uncertain if the concentrations measured in the batch leaching test are those representative of point A, B or C (as presented in Figure 2-2). As a result these tests still do not provide a clear indication of the C_L . There still is a need to measure C_L as accurately as possible using a leaching tests with at a liquid-to-solid ratio that better represents rain fall through the solid waste, like a lysimeter test as previously discussed.

2.3.4 Lysimeter Leaching Tests Approach

An alternative to batch tests, column or lysimeter tests permit the simulation of waste leachate concentrations closer to that encountered in actual land-applied waste conditions. Lysimeter studies examine contaminant mobility from solid wastes at a low liquid-to-solid ratio. These tests usually involve the incremental application of a small volume of leaching solution to the solid waste (Hjelmar 1990; Kedziorek et al. 1998; Obiath 1989). Unlike batch leaching tests, which are conducted at a constant liquid-to-solid ratio, the liquid-to-solid ratio of lysimeter tests changes as more extraction fluid is applied.

Although no accepted standardized protocol is available for these test methods in the US, some states accept results from lysimeter studies on a case-by-case basis as they generate more concentrated leachate that is indicative of pore water concentration. Since the liquid-to-solid ratios in lysimeter tests are relatively small (< 1) and contact time between the leaching fluid and the waste is relatively long, it could be easily argued that lysimeter leachate concentrations are indicative of pore water concentrations (point A on Figure 2-2). By applying the appropriate DAF to these concentrations an estimate of actual groundwater concentrations C_{RC} could be achieved. Unlike batch leaching tests, lysimeter leaching tests require relatively long time to conduct and are more expensive.

2.5 Characteristics of Wood Ash and Tire Ash

The co-combustion of vehicle scrap tires and waste wood is a management option for these solid wastes that produces valuable energy. A byproduct of the combustion process is wood and tire (WT) ash. While landfills are the most common ash disposal option (Campbell et al. 1990), economic factors and the desire for resource conservation have spurred interest in the beneficial use of ash (Lerner and Utzinger 1986; Naylor and

Schmidt 1986; Schreurs et al. 2000; Vance 1996). For example, wood ash from the combustion of clean wood has historically been used as a liming agent (soda ash) and as a source of plant nutrients (Campbell et al. 1990). The combustion of other wood waste, such as that from construction and demolition (C&D) debris, may contain impurities such as heavy metals from preservative treated wood (Tolaymat et al. 2000) and wood painted with lead-based paint (Beebe and England 1998). These impurities then cause an increase in metal concentrations in the resulting ash. In a similar manner, the addition of tires to the combustion of the wood waste can change the composition of the ash and has the potential to increase impurities.

Historically, wood ash has been beneficially used as road-base, and fill material (Campbell et al. 1990; Miller and Rahe 1990; Schreurs et al. 2000). The most common beneficial use option for wood ash, however, is as an agricultural amendment. Wood ash serves as a source of potassium, calcium, phosphorous and is an effective liming agent (Campbell et al. 1990; Etiegni et al. 1991; Lerner and Utzinger 1986). Calcium oxide (CaO) is the prevalent form of calcium produced during the combustion of wood, but interactions with water and carbon dioxide transform it into calcium carbonate (CaCO_3) over time (Campbell et al. 1990; Etiegni and Campbell 1991). A study examining the effect of wood ash on a northern European pine forest soil concluded that the buffering capacity of wood ash was similar to conventional lime (Bramryd and Fransman 1995). Other studies concluded that the availability of potassium in wood ash was similar to that of fertilizers (Erich 1991; Naylor and Schmidt 1986; Ohno 1992).

The chemical and physical characteristics of wood ash vary depending on the species of wood combusted and incineration conditions (Campbell et al. 1990; Misra et

al. 1993). Since heavy metals become concentrated in the ash, appropriate management practices must be taken to minimize the environmental impact during ash disposal or beneficial use. Metal concentrations (mg/kg) in ash from the combustion of clean wood have been thoroughly characterized (Bramryd and Fransman 1995; Erich 1991; Etiegni et al. 1991; Lerner and Utzinger 1986; Naylor and Schmidt 1986; Ohno 1992; Vance 1996) and typical concentrations are summarized in Table 2-5.

Table 2-5: Average Wood Ash Metal Concentration

Metal	Units	Average Concentration (mg/kg)
Al ^{a,b}	g/kg	14.5 ± 7
Ca ^{a,b}		171 ± 80
Fe ^{a,b}		9.31 ± 5
K ^{a,b}		25.8 ± 7
Mg ^{a,b}		12.2 ± 5
Na ^{a,b}		1.8 ± 1.3
As ^c	mg/kg	23.2 ± 20
Cd ^c		5.0 ± 5
Cr ^c		39.0 ± 30
Co ^c		8.7 ± 5
Cu ^c		75.3 ± 45
Pb ^c		65.6 ± 40
Mn ^c		4370 ± 2700
Ni ^c		23.5 ± 20
Zn ^c		443 ± 400

a (Muse and Mitchell 1995)

b (Campbell et al. 1990)

c (Someswar 1996)

A smaller number of studies have evaluated the leachable metal concentrations from wood ash (Erich 1991; Xiao et al. 1999). Most leaching research on wood ash has been conducted to determine available plant nutrients for agricultural purposes, not to evaluate potential environmental impacts of heavy metal leaching to groundwater (Erich 1991). Xiao et al. (1999) demonstrated the potential for heavy metals such as lead to leach from wood ash when exposed to deionized water. However, Campbell et al. (1990)

concluded that since wood ash is not a hazardous waste and nutrients are readily available, it could be used as an agricultural additive.

Although the US EPA estimated that approximately 57% of the 270 million scrap tires generated in 1999 were incinerated (EPA 1999a), the characterization of scrap tire ash is limited since it is rare for scrap tires to be used as the sole fuel source in a combustion process. For this reason, the beneficial use of specifically scrap tire ash has not been examined by researchers. Tires are made from vulcanized rubber reinforced with a metallic (primarily steel) mesh. Most of the ash remaining after scrap tire combustion is comprised of the metallic mesh (Rodriguez et al. 2001). Levie et al. (1995) reported that scrap tire bottom ash contains 96% by weight iron, while the scrap tire fly ash contains 51% zinc by weight. Both ash types also contain trace levels of other elements like arsenic, chromium, cadmium and lead (see Table 2-6). The leaching of

Table 2-6: Metal Concentration In Fly and Bottom Shredded Tire Ash^a

Metal	Fly ash (mg/kg)	Bottom ash (mg/kg)
As	200	10
Al	7,600	20,600
Pb	2,200	10
Cr	300	5,230
Cd	500	10
Mn	NM	4,160
Zn	51.48%	1,060
Fe	63,300	96.2%

a (Levie et al. 1995)

NM not measured

metals from waste tire ash was not assessed by this same study. While some characterization data are available for scrap tire ash and wood ash individually, one cannot necessarily make definitive conclusions about the characteristics of WT ash. Even though each waste will contribute (by weight) constituents based upon the

characterization of each of the ashes (wood or tires) described above, the interaction of these different constituents is unknown until they are physically tested together.

CHAPTER 3 MATERIALS AND METHODS

As described previously, ash from a waste-to-energy facility that combusts wood and tires was tested for this research. The ash was selected because 1) historical records indicate that it was non-hazardous, 2) there was an expressed interest from the ash generator to reuse the material through land application and 3) the ash did have potential to contain low level of trace pollutants that could limit reuse. In addition, the leachate at the landfill where the waste was co-disposed with MSW contained elevated level of arsenic, with the ash being a key suspect for contributing arsenic. This section presents the methods used to chemically characterize the inorganic content of the ash. The standardized leaching experiments, as well as those designed to meet the project's objectives are described. The quality control procedures, statistical procedures and the contaminant transport model approaches are also described.

3.1 Wood and Tire (WT) Ash Sample Collection

The ash for this study was generated at a waste-to-energy (WTE) facility in Florida that combusts waste wood amended with shredded scrap vehicle tires. The facility accepts wood from land clearing and construction and demolition (C&D) activities, as well as some industrial wood sources and yard trash. Shredded and whole (without rims) scrap tires are also accepted by the facility. Larger pieces of waste wood and tires are shredded to less than 3 and 6 cm, respectively, before combustion. Shredded tires comprise up to 10% of the fuel stream. The facility generates 150 to 300 tons of WT ash daily that is co-disposed with MSW at a neighboring sanitary landfill. The WT ash

sampling consisted of a mixture of bottom and fly ash as well as air pollution control (APC) residues. Initial WT ash samples were collected from eight ash truckloads during one sampling event in October 2000. Each truck emptied its load near the working face of the landfill and a front-end loader mixed the WT ash before sample collection. Eight samples (one from each truckload) were collected in 38-L (10-gal) PVC containers and transported to the laboratory. Six subsequent WT ash samples (1-L) were collected after the initial sampling event throughout 2001.

Each of the initial eight samples was mixed again in the lab to promote homogeneity of the samples. Approximately equal amounts (see Appendix A) of the eight samples were used to generate a composite sample; this composite sample was used for further characterization of the WT ash. The additional six samples collected (during 2001) were analyzed for total metals. These samples were also analyzed for leachable metal content using the synthetic precipitation leaching procedure (SPLP). These subsequent samples were collected to allow for better examination of metal variability over time.

3.2 Morphological Analysis

The morphological analyses were conducted at the Material Science and Engineering Research Center at the University of Florida. The Center is a materials characterization and analysis facility established to provide analytical support for Florida's scientific and engineering community. It provides service to the University of Florida, the state university system (SUS), and the industrial and commercial community. The analyses were conducted on the composite WT ash sample. An unprocessed (intact) WT ash sample was used for scanning electron microscope (SEM) analysis. SEM analyzes the morphology of the ash samples using a SEM Jeol JSM 35CF (Sollentuna,

Sweden) with a typical penetration depth of 10 Å. Another WT ash sample was manually crushed and then analyzed using X-ray diffraction (XRD) using a Philips APD 3720 (New York, US). XRD is a bulk analytical technique with a penetration of 1 µm. This technique is used to identify crystalline phases in the ash sample.

3.3 Measurements of Total Extractable Metals

Total extractable metal concentration was obtained by conducting a hot acid digestion following the EPA's SW-846 3050B (EPA 1996b). This method estimates the maximum amount of extractable metal in the waste of interest. Between 1 and 2 g (Appendix B), corrected for moisture content (ash moisture content presented are in Appendix C), of WT ash were weighed into a flask. The digestion procedure was a hot plate open vessel method requiring the use of nitric acid, hydrogen peroxide and hydrochloric acid (for inductively coupled plasma (ICP) analysis). The samples were refluxed on a hot plate for a period ranging from 2 to 8 hours, depending on the sample type. This type of analysis gives an indication of the maximum extractable metals but does not give an indication of leachable metal content.

3.4 Batch Leaching Tests

The regulatory batch leaching tests performed included the toxicity characteristic leaching procedure (TCLP) (EPA 1996b), the SPLP (EPA 1996b), the multiple extraction procedure (MEP) (EPA 1996b) and California's waste extraction test (WET) (CCR 1998). While the TCLP is the batch test that is required by the Resource Conservation and Recovery Act (RCRA) for hazardous waste characterization, the state of California also uses the WET for this purpose. Both the TCLP and the WET examine metal leachability from solid wastes in MSW landfill environments and thus utilize organic acids. The SPLP uses a slightly acidic extraction solution to assess the impact of acid

rain fall on metal leachability from solid wastes. The MEP examines long term metal leachability by utilizing both an organic acid extraction solution followed a slightly acidic rainwater extraction solution. Since these tests use varying extraction solution to examine metal leachability under different environmental conditions, and since field pH and liquid-to-solid ratio may be considerably different than those measured under the standardized batch leaching tests, it was necessary to conduct specific batch tests targeting different factors (pH, contact time, and liquid to-solid ratio) that affect metal leachability from solid wastes.

3.4.1 Toxicity Characteristic Leaching procedure (TCLP)

Depending on the alkalinity of the waste, the TCLP calls for one of two extraction fluids. Extraction fluid #1 is prepared by diluting a mixture of 11.4-mL acetic acid (CH_3COOH) and 128.6-mL of 1N sodium hydroxide (NaOH) to two liters using reagent water. The final pH of the solution must be 4.93 ± 0.05 . For highly alkaline materials (such as wood ash), extraction fluid #2 is used. This extraction solution is prepared by diluting 11.4-mL acetic acid (CH_3COOH) to two liters using reagent water. The extraction fluid pH must be 2.80 ± 0.05 . The extraction solution, in this case solution #2, was added to a representative ash sample to achieve a liquid-to-solid ratio of 20:1. The mixture was rotated for 18 ± 2 hours. After rotation, the final pH was measured and the slurry was filtered using a $0.7 \mu\text{m}$ Whatman glass fiber filter. The filtrate was collected in a plastic bottle and nitric acid was added until the pH of the solution was below 2 (EPA 1996b).

3.4.2 Synthetic precipitation leaching procedure (SPLP)

The SPLP utilizes two inorganic acids (nitric and sulfuric acid) to simulate acidic rainwater. East of the Mississippi River, the fluid is slightly acidic at a $\text{pH } 4.20 \pm 0.05$,

which reflects the impact of air pollution from heavy industrialization and coal utilization. An extraction solution with a pH of 5.0 ± 0.05 is used west of the Mississippi, reflecting less industrialization and smaller population densities. The SPLP leaching solution was prepared by carefully mixing 60-g of sulfuric acid with 40-g of nitric acid. The SPLP extraction fluid was prepared by adding between 0.4 and 0.5-ml of the sulfuric acid / nitric acid mixture to a 2-L volumetric flask and diluting it to volume with reagent deionized water. The resultant pH was 4.20 ± 0.05 . From this point on the test follows the same steps as the TCLP (EPA 1996b).

3.4.3 Waste Extraction Test (WET)

The WET extraction solution is prepared by titrating a 0.2-M citric acid solution with 4.0-N NaOH to a pH of 5.0 ± 0.1 . One liter of this fluid is added to a 100-g sample and rotated for a period of 48 hours. From this point on the test follows the same steps as the TCLP (CCR 1998).

3.4.4 Multiple Extraction Procedure (MEP)

The extraction fluids used in the MEP consist of both an organic acid (acetic acid) used to simulate MSW leachate and an inorganic acid mixture (nitric and sulfuric acids) designed to simulate acid rain. Extraction was initially performed on 60-g of each sample using an acetic acid solution. After the 24-hour rotation period and filtration of the leachate, seven additional extractions were performed on the solids captured on the filter. The solution used for these extractions was a nitric/sulfuric acid solution (i.e., synthetic rain) with a pH of 3.0 ± 0.2 . During each subsequent extraction, the synthetic rain extraction fluid was added to the waste at a 20:1 ratio, the samples were rotated for 24 hours. Filtration and preservation followed the same method as the TCLP (EPA 1996b).

3.4.5 Additional Leaching Tests

In order to evaluate the effects of liquid-to-solid ratio on heavy metal leachability, the SPLP extraction test was modified to achieve different liquid-to-solid ratios. These ratios were attained by adding an appropriate volume of synthetic acid rainwater to 100-g of ash to generate 3, 5, 10, 20 and 50 (L/kg) liquid-to-solid ratios. To achieve higher liquid-to-solid ratios (100, 200, 300, 400, and 500 L/kg), an appropriate volume of synthetic rainwater was added to 10-g of ash. Extraction time was held constant at 18 hours. The effects of changes in extraction pH on heavy metal leachability were also examined. While holding extraction time at 18 hours and the liquid-to-solid ratio at 20:1, the pH of the ash extraction fluid slurry was varied.

Different pH values were achieved by adding 150-mL of deionized water to 10-g of ash. The slurry was mixed and the pH was monitored continuously with a pH probe. Either diluted nitric acid or sodium hydroxide was added accordingly to maintain the desired pH. The pH was varied from 1 to 13 pH units. Once the pH stabilized at the predetermined set point, enough reagent water was added to achieve a liquid-to-solid ratio of 20:1.

To address the effects of contact time on heavy metal leachability, the SPLP was followed; however, the extraction time was changed to achieve different contact times. Contact time varied as follows: 1, 2, 4, 8, 12 hours, then 1, 2, 4, 5, 6 and 7 days. Again, the liquid-to-solid ratio was held constant at 20:1. Filtration and preservation for these extraction tests followed the procedure presented earlier for the TCLP.

3.5 Lysimeter Leaching Tests

Lysimeter studies were conducted using stainless steel leaching columns, described previously by Brantley and Townsend (1999). The experiment included three different

types of lysimeters with each type in triplicate for a total of 9 lysimeters (Figure 3-1). Type 1 lysimeter contained 0.3 m (1 ft), type 2 contained 0.61 m (2 ft) and type 3 contained 0.91 m (3 ft) of ash. Each of the eight ash samples were mixed in the laboratory to promote homogeneity. The eight ash samples were placed in layers to achieve the desired depth of ash for each lysimeter type as presented in Figure 3-1. For example, in type 1 lysimeters, each sample contributed a layer of ash that is 3.8 cm (1.5 inch) deep for a total of 0.3 m (1ft) of ash. For lysimeter types 2 and 3, each ash sample contributed a layer that is 7.6 and 11.5 cm (3 and 4.5 inches) respectively. A tenth lysimeter filled with sand was used as control. As it was placed each ash layer was compacted to a density of 1300 kg/m^3 (80 pcf). The hydraulic conductivity was measured to be 10^{-3} cm/sec . Simulated acid rainwater (SPLP extraction fluid with a pH of 4.2 ± 0.05) was percolated through the columns at a rate of 5 ml every 30 minutes ($2.8\text{E-}3 \text{ cm/sec}$) using a computer controlled peristaltic pump. The ash samples were leached over a period of 420 days. The leachate was drained every 4 days for the first two months, every week for the following two months and then once a month for remainder of the experiment.

3.6 Leachate Testing for General Leachate Quality Parameters

The methods employed were adapted either from standardized methods (APHA, 1995) or the US EPA SW-846 (US EPA 1996b). The pH, oxidation-reduction potential (ORP), conductivity, total dissolved solids (TDS) and alkalinity methods are presented in Table 3-1. Leachate samples preserved with sulfuric acid were analyzed for non-purgable organic carbon (NPOC) and chemical oxygen demand (COD) following methods presented in Table 3-1.

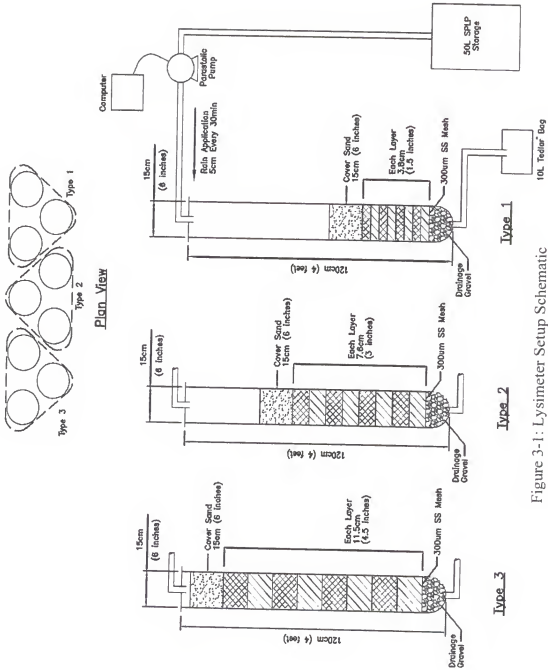


Figure 3-1: Lysimeter Setup Schematic

Table 3-1: Analytical Methods for Leachate Analysis

Parameter	Method	Instrument
pH	SM ^a 4500-H ⁺	Fisher Scientific Accumet® Model 20 equipped with an Orion probe 9156
Oxidation Reduction Potential (ORP)	SM ^a 2580	Fisher Scientific Accumet® Model 20 equipped with an Orion probe 91798N
Conductivity	SM ^a 2510B	Hanna Instruments HI90330 Multi Range
Total Dissolved Solids (TDS)	SM ^a 2540C	None
Alkalinity	SM ^a 2320B	Fisher Scientific Accumet® Model
Chemical Oxygen Demand (COD)	SM ^a 5220C	Hach DR-4000 Spectrophotometer
Nonpurgable Organic Carbon (NPOC)	SM ^a 5310B	Romemount Analysitcal Dohrman DC-190
Anions	SW-846 method 9056 ^b	Dionex DX-400
Cations	EPA method 300.7	Environ 60 ICP-AES
Metals	SW-846 method 3010	Environ 60 ICP-AES (Al, Ba, Be, Fe, Cd, Co, Mn, Zn, V)
	SW-846 method 3020	Perkin Elmer GFAA 5100 (As, Cu, Cr, Pb, Ni, Se)

Filtered leachate samples were analyzed for anions using an ion chromatograph (Dionex DR-400) as presented in Table 3-1. Filtered leachate samples were analyzed for cations using inductively coupled plasma atomic emission spectroscopy (Thermo Jerral Ash Environ 60 ICP-AES).

3.7 Analysis of Metals and Metalloids

Liquid samples were digested according to US EPA SW-846 method 3020 for inductively coupled plasma analysis or 3010 for graphite furnace analysis (EPA 1996b). Liquid samples that are analyzed for arsenic were digested using US EPA SW- 846 method 7060A (EPA 1996b). These methods are open vessel methods that require the addition of concentrated nitric acid to a representative 100 ml sample. However, method

3020 requires an addition of hydrochloric acid at the final step of the digestion.

Hydrochloric acid was not used for the GFAA analysis since it causes interferences during the atomization process (EPA 1996b).

The total metal analysis for all metals except arsenic was conducted using a Thermo Jarrel Ash ICP spectrophotometer model Evrion 60 (Franklin, US). Perkin-Elmer graphite furnace atomic adsorption 5100 (Boston, US) with Zeeman background correction was used to analyze total arsenic, and leachable arsenic, lead, copper, chromium and nickel concentrations. Other leachable metals were analyzed using the previously mentioned ICP-AES. Instrumentation Detection Limits are presented in Appendix D and quality assurance/quality control in Appendix E.

3.8 Quality Assurance and Quality Control

To implement quality assurance and quality control practices in the field all sampling equipment and containers were cleaned and prepared according to the established US EPA procedures (EPA 2000). In the laboratory, analysis blanks, duplicates and calibration check samples were analyzed as appropriate. For total metal analysis every digestion event (approximately 36 samples) a blank sample, a duplicate sample, a triplicate sample, a sample spike and sample spike duplicate were included in the digestion. All regulatory batch-leaching tests were conducted in triplicates as a result no duplicate samples were extracted. However, during the digestion process, every digestion event (approximately 36 samples) included a blank sample, a duplicate sample, a triplicate sample, a sample spike and sample spike duplicate in the digestion process.

Contact time batch leaching tests and liquid-to-solid ratio batch leaching tests were conducted in triplicate. Thus, no duplicate samples were extracted and digestion spikes and blanks were included like with the regulatory batch leaching tests. No duplicate

samples were extracted on pH static leaching test. However, duplicate digested samples and blanks were included as in the regulatory batch leaching test. Sample holding times and digestion volumes were followed as prescribed by the US EPA (EPA 2000).

3.9 Statistical Analysis

For comparison of the different extraction tests, statistical analysis suggested by the US EPA was followed (EPA 2000). Summary of the two statistical methods utilized in this dissertation are presented in the following sections. For statistical analysis, half of the detection limit was used for concentrations that were below the detection limit. The first statistical tool used is Student's two-sample t-Test, which examines the statistical similarities between two population means. The student t-Test was carried out to compare the extractability of different regulatory batch leaching tests, the effect of depth on leachate quality in the lysimeter tests, and on comparing lysimeter results to batch leaching tests results. In general, at every even where two means were statistically compared the student t-Test was used. The second tool utilized is the upper 95 percent confidence limit on the mean UCL_{95} , which is a conservative method to estimate the mean of a population. The UCL_{95} on the mean was only used to compare extraction test results (lysimeter, batch, total extractable metals) with risk-based regulatory limits.

3.9.1 Student's Two-Sample t-Test

The principal assumption required for the two-sample t-test is that a random sample of size m ($X_1, X_2, X_3, \dots, X_m$) is drawn from population 1, and an independent random sample of size n ($Y_1, Y_2, Y_3, \dots, Y_n$) is drawn from population 2. The first step of the test is to calculate the sample mean \bar{X} (using equation 3-1) and sample variance S_x^2 (using equation 3-2) for sample 1 and compute the sample mean \bar{Y} and the sample variance S_y^2 for sample 2.

$$\bar{X} = \frac{\sum_{i=1}^m x_i}{m} \quad (3-1)$$

Where \bar{X} is the arithmetic mean of the m samples, x_i is the single sample value, and m is the number of samples.

$$S_x^2 = \frac{\sum_{i=1}^m (x_i - \bar{X}_i)^2}{m-1} \quad (3-2)$$

Where S_x^2 is the arithmetic variance of the m samples, x_i is the single sample value, and \bar{X} is the arithmetic mean of the m samples.

The next step is to determine if the variance of the two populations are equal. If the variance is not equal use the section 3.9.1.1 to compare the population means. If the variance is equal then use section 3.9.1.2 to compare the population means.

3.9.1.1 Unequal variances

The first step is to compute S_{NE} using equation 3-4

$$S_{NE} = \sqrt{\frac{S_x^2}{m} + \frac{S_y^2}{n}} \quad (3-4)$$

Where S_x^2 is the arithmetic mean of the m and S_y^2 is the arithmetic mean of the n samples. Then calculate t using equation 3-5.

$$t = \frac{\bar{X} - \bar{Y}}{S_{NE}} \quad (3-5)$$

Calculate the degrees of freedom using equation 3-6.

$$f = \frac{\left[\frac{S_x^2}{m} + \frac{S_y^2}{n} \right]^2}{\frac{S_x^4}{m^2(m-1)} + \frac{S_y^4}{n^2(n-1)}} \quad (3-6)$$

Look up critical value $t_{1-\alpha}$. If $t > t_{1-\alpha}$ then the population means are different, if $t < t_{1-\alpha}$ then there is not enough evidence to the two population means are different.

3.9.1.2 Equal variances

The first step is to compute S_E using equation 3-7

$$S_E = \sqrt{\frac{(m-1)S_X^2 + (n-1)S_Y^2}{(m-1) + (n-1)}} \quad (3-7)$$

Calculate t using equation 3-8

$$t = \frac{\bar{X} - \bar{Y}}{S_E \sqrt{\frac{1}{n} + \frac{1}{m}}} \quad (3-8)$$

Calculate the degrees of freedom using equation 3-9

$$f = (m + n - 2) \quad (3-9)$$

Look up critical value $t_{1-\alpha}$. If $t > t_{1-\alpha}$ then the population means are different, if $t < t_{1-\alpha}$ then there is not enough evidence to the two population means are different.

3.9.2 Upper Confidence Limit (UCL)

The 95 percent UCL of a mean is defined as a value that, when calculated repeatedly for randomly drawn subset of site data, equals or exceeds the true mean 95 percent of the time. The 95 percent UCL of the arithmetic mean concentration is used as the mean concentration, because it is not possible to know the true mean of the population. The 95 percent UCL, therefore, accounts for uncertainties from limited sampling data. As sampling data become less limited at a site, uncertainties decrease; the UCL moves closer to the true mean, and exposure evaluations using either the mean or the UCL produce similar results. In general, the UCL_{95} on the mean is a conservative estimate of the mean. The UCL can be calculated using equation 3-10.

$$UCL = \bar{X} + t \left(\frac{s}{\sqrt{n}} \right) \quad (3-10)$$

Where UCL is the upper confidence limit, \bar{X} is the mean, s is the standard deviation, t is the student-t static, n is the number of samples.

3.10 Transport Model (MYGRT)

As stated in chapter 2, the degree of dilution occurring between the solid waste application site and the point of compliance is unclear. The US EPA used computer programs results to estimate a dilution factor of 20 for most sites that are smaller than 0.5 acres (see section 2.4.1). However, it is still possible to calculate a site specific dilution factor using contaminant transport models like (MYGRT). Contaminant concentrations in lysimeter leaching tests were used as input values to model ground water contamination from the land application of WT ash. The hypothetical scenario examines the migration of contaminant from a land application area of 200 m long and wide over a period of 200 years. The computer-modeling program MYGRT was used for the simulation.

3.10.1 Land Application Scenario

The hypothetical scenario presented here simulates the migration of contaminants from a land-applied WT ash lift. A schematic of the scenario is presented in Figure 3-2. The unlined ash lift is land-applied at a location in central Florida (Orlando). The ash lift is 200 m long (measured parallel to the ground water flow direction),

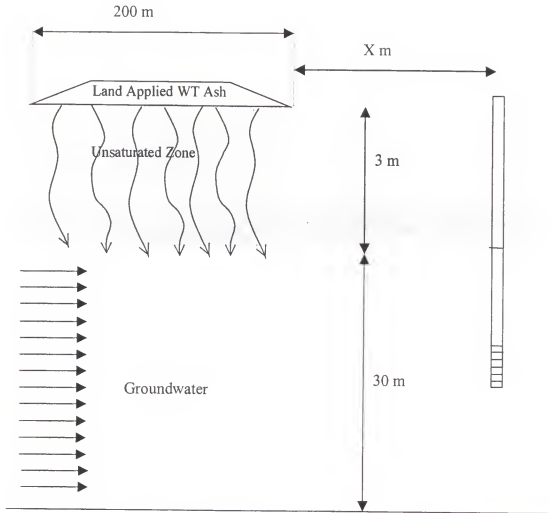


Figure 3-2: Proposed WT Ash Land Application Scenario

and 200 m wide (measured perpendicular to ground water flow direction). Observation wells are located down gradient of the pond and are screened in the aquifer between 1 and 6 m below the water table. The computer program models contamination in the surficial aquifer, which has an unsaturated zone thickness of 10 m. Underlying the unsaturated layer is a 30 m thick aquifer, with a hydraulic conductivity of 5 m/day. Leachate infiltrates into the unsaturated zone at a rate of 0.33 m/year and a 0.0015 gradient. The estimated life span of the ash lift is 200 years. Specific program input parameters are presented in Table 3-2.

Table 3-2: MYGRT Modeling Input Parameters

Description	Value
Unsaturated Infiltration Rate	0.4 m/yr
Unsaturated Moisture Content	0.2
Unsaturated dispersion Coefficient	1.98 m ² /yr
Depth to Water Table	3 m
Width of source	200 m
Length of Source	200 m
Aquifer Depth	30 m
Seepage Velocity	0.0158
Hydraulic Gradient	0.0015 m/yr
Hydraulic Conductivity	2000 m/yr
Upper Observation Well Screen	1
Lower Observation Well Screen	6
Background Contaminant Concentration	0 mg/L
Aquifer pH	6.8
K _d value at pH 6.8 for Zn	62 L/kg
K _d value at pH 6.8 for Al	49 L/kg
K _d value at pH 6.8 for Pb	2000 L/kg

This scenario is modeled as a contaminant migrating down through the one-dimensional unsaturated soil, mixing with the underlying water table and then moving down gradient through the three-dimensional finite aquifer. Equations governing the model are presented in the next section.

3.10.2 Overview of The Modeling Program (MYGRT)

MYGRT Version 3 is used to simulate single species solute migration through several pathways, starting from surface sources and migrating downward through the unsaturated soil layer, mixing with the underlying ground water, and then migrating horizontally down gradient through the aquifer. Each of these pathway models may be simulated individually or as an interconnected system of models. Upon defining source concentration as a function of time (piecewise constant) and soil and chemical parameters of the unsaturated/saturated layers, the model predicts the solute concentration as a function of time at any specified location along its pathway.

The MYGRT program is a collection of 22 models based on analytical solutions to the one, two and three dimensional mass transport equations using the integral transform technique. The integral transform method starts with the partial differential equation, boundary conditions, and initial conditions of a transport problem and proceeds to an exact solution of it. The aquifer can have a finite or semi-infinite thickness. Up to one, two, or three dimensions can be simulated in the saturated zone. For this scenario two dimensional modeling was used.

To simulate our use scenario, a two-step algorithm was used to generate the times at which the boundary condition was computed by MYGRT for the saturated layer. MYGRT utilizes contaminant transport equation (3-11) with retardation and advection only in one dimension to generate equation 3-12.

$$D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} + D_z \frac{\partial^2 C}{\partial z^2} - V_x \frac{\partial C}{\partial X} - \lambda RC = R \frac{\partial C}{\partial t} \quad (3-11)$$

$$D_x \frac{\partial^2 C}{\partial x^2} - V_x \frac{\partial C}{\partial X} - \lambda RC = R \frac{\partial C}{\partial t} \quad (3-12)$$

Where D_x , D_y , and D_z are the dispersion coefficients in the x, y and z dimensions respectively, V_x is the velocity in the x direction, R is the retardation factor, λ is a decay constant and C is the concentration.

MYGRT uses analytical method to solve the partial differential equation presented earlier (equation 3-12). The model then predicts the resultant solute concentration at the bottom of the unsaturated layer. These concentrations are then curve-fitted with linear functions versus time. At the end of this process, the boundary condition versus time for the saturated layer is described by linear functions. For this scenario, the saturated zone

was assumed to be a finite two dimensional aquifer with the governing equation as the one presented in equation 3-11.

CHAPTER 4

CHARACTERIZATION OF TOTAL AND LEACHABLE METAL CONTENT OF WOOD AND TIRE ASH

WT ash total metal content (mg/kg) (using a solid digestion procedure) and leachable metal content (mg/L) (using regulatory batch leaching tests) are presented in the following sections. The metal concentrations measured in the WT ash are also statistically compared with metal concentrations of virgin wood ash from the literature to examine the effects of tires. Besides regulatory batch leaching tests, batch leaching tests examining the effects of pH, liquid-to-solid ratio and contact time are also presented.

4.1 Total Metals

Acid digestion was performed on the initial composite WT ash sample (analyzed in triplicate) and the six individual ash samples collected on different dates. Raw metal concentration data are presented in Appendix F. Seventeen metals were detected (aluminum, arsenic, barium, calcium, cadmium, cobalt, chromium, copper, vanadium, iron, potassium, lead, magnesium, manganese, sodium, nickel and zinc). All 17 metals were measured at concentrations above their respective detection limits. Table 4-1 presents the average metal concentrations of the triplicate composite samples and the standard deviation. Calcium (223 g/kg) was the most abundant metal detected in the ash. Of the more toxic metals, lead had the highest concentration (63.1 mg/kg). Arsenic was also detected at a somewhat elevated concentration (37.2 mg/kg).

Table 4-1: Total Metal Concentration in WT Ash

Metal	Units	Metal Concentration (mg/kg)							Form
		10/15/00 ^{a,b} Mean \pm SD	1/24/01 ^c	1/31/01 ^c	2/8/01 ^c	4/6/01 ^c	4/12/01 ^c	9/12/01 ^c	
Al		3.94 \pm 0.7	2.09	2.01	3.28	2.43	2.51	2.90	AlPO ₄
Ca		223 \pm 50	107	89.2	75.8	124	149	89.7	CaO, CaCO ₃
Fe		34.7 \pm 5	74.4	19.1	9.50	10.9	6.83	10.5	Fe ₃ PO ₇
K	mg/kg	6.67 \pm 0.8	4.80	3.86	3.56	3.57	4.18	3.99	K ₂ SO ₄ , KClO ₃
Mg		5.42 \pm 1	3.06	2.92	3.29	3.51	3.41	2.90	Al ₃ Mg ₂
Na		1.80 \pm 0.02	0.975	1.06	1.10	0.983	0.991	1.06	ND
Zn		18.2 \pm 3	11.2	8.36	7.69	11.4	5.68	8.78	ZnP ₄ O ₁₁ , ZnSO ₄
As		37.2 \pm 6	35.8	36.8	38.9	32.1	34.8	36.7	ND
Ba		39.3 \pm 7	25.3	22.2	33.7	32.4	30.4	34.0	ND
Cd		2.71 \pm 0.5	2.28	1.21	1.27	1.73	1.51	1.29	ND
Cr		46.3 \pm 5	36.5	25.2	27.4	34.2	23.1	34.7	ND
Co		129 \pm 30	73.2	68.9	56.8	86.8	40.5	59.0	ND
Cu	mg/kg	162 \pm 30	197	94.8	71.4	94.5	63.2	90.3	ND
Pb		63.1 \pm 10	57.3	37.4	37.9	45.4	38.0	50.0	ND
Mn		307 \pm 80	461	157	228	337	419	145	ND
Ni		16.7 \pm 4	15.0	8.36	7.34	7.89	5.93	7.24	ND
V		5.49 \pm 2	11.2	5.64	7.61	5.10	4.13	5.87	ND

NID=Not Detected

^a Composite ash sample of the initial 8 ash loads^b Concentration of one sample analyzed in triplicate^c Concentration of only one sample analysis

4.1.1 Variability of Total Metal Content

Variability in metal concentrations between the triplicate analyses was observed but not unexpected. For example, the concentration of manganese (307 mg/kg) was associated with 80 mg/kg standard deviation which is almost a third of the manganese concentration. This variability can be caused by the heterogeneity of the WT ash samples coupled with the small mass (2-g) used in the digestion method. Variability in metal content over time (the 6 samples collected during 2001) was also observed and is presented in Table 4-1. The initial concentration of zinc, for example, was 18.2 g/kg, that concentration dropped to a minimum of 5.68 g/kg (on 4/12/2001) which is about a third of the initial concentration. The concentration of calcium ranged from a 223 g/kg (on 10/15/2000) to 75.8 g/kg (on 2/8/2001). The observed variability over time was larger than that observed among the triplicate composite samples. While inconsistency between triplicate samples reflects variability within the sampling and analysis methods, variability over time may be related to changes in the combustion process, feedstock composition and operational changes over time (Bramryd and Fransman 1995; Buchholz and Landsberger 1995a; Glordano et al. 1983).

4.1.2 Morphological Analysis

The crystalline forms of the metals were examined using SEM and XRD. While SEM analysis indicates whether a particular crystalline form is present in the ash, XRD gives the actual composition of that crystalline form. Because of the high concentrations of some elements relative to others, the analysis was unable to evaluate forms of metals with only trace concentrations. The results presented in Table 4-1 indicate crystalline forms consistent with those reported in the literature to be present in wood ash (Misra et al. 1993; Steenari and Lindquist 1999). Calcium, the most abundant element in WT ash,

is primarily in CaO form. The presence of CaO gives the WT ash a buffering capacity similar to that of wood ash. Aluminum, iron and zinc were also detected in oxide forms.

4.1.3 Comparison to Wood Ash Metal Content

As mentioned earlier, there were no data available in the literature on WT ash, thus metal concentrations in WT ash were compared to wood ash. To evaluate the effects of the co-incineration of tires with wood on the resulting ash, the average metal concentrations of WT ash for all seven samples (the composite collected during 2000 and the 6 samples collected during 2001) were statistically compared (using the student T-test) to typical wood ash concentrations reported in the literature (Table 2-5). The wood ash concentrations presented in Table 2-5 are average concentrations from a variety of sources and thus represent a wide range of wood species and combustion conditions. The concentrations of iron, zinc, copper, cobalt and sodium in the WT ash were higher ($\alpha = 0.05$) than in wood ash alone. The concentrations of nickel, cadmium, potassium, magnesium, manganese and aluminum were lower ($\alpha = 0.05$) in the WT ash than in the wood ash. The concentrations of arsenic, chromium, lead and calcium in the WT ash were similar ($\alpha = 0.05$) to that present in wood ash alone. Studies conducted by Steenari and Lidquist (1999) suggested that the co-incineration of other fuels with wood may lower the calcium content of the resulting ash. It is noted, however, that lime is used in the air pollution control system at the facility, and therefore might have led to the increased calcium concentrations.

It is apparent that the co-combustion of shredded tires with wood resulted in an enhancement in concentrations of the metals most abundant in tire ash such as iron and zinc. Although research conducted by Levie et al. (1995) concluded that tire ash contains

some of the more toxic metals like arsenic (200 mg/kg in fly ash and 10 mg/kg in bottom ash) and lead (2,200 mg/kg in fly ash and 10 mg/kg in bottom ash), the addition of shredded tires did not increase the concentrations of these metals in the WT ash. This might be caused by a combination of the wide range of background metal concentrations in wood ash samples (Table 2-5) and the small fraction of the fuel stream comprised of shredded tires (10% or less).

The use of wood recovered from C&D debris did not lead to an increase the concentrations of metals like arsenic and chromium. Studies have found that the ash from the combustion of C&D debris recovered wood often contains elevated levels of arsenic, chromium and copper as a result of pressure treated wood (Solo-Gabriele et al. 2002; Tolaymat et al. 2000). Chromated copper arsenate (CCA)-treated wood, which may be present in recovered C&D debris wood, contains high concentrations of these three metals (Tolaymat et al. 2000). Therefore, the presence of small amounts of CCA-treated wood could impact the overall concentrations of these metals in the C&D debris wood ash. The concentration of arsenic, for example, in ash from the combustion of C&D debris wood collected in Florida was found to range from 750 to 2250 mg/kg (Solo-Gabriele et al. 2002). The arsenic concentration of WT ash (37.2 mg/kg) was lower than expected from combusting C&D debris wood suggesting that the fraction of C&D debris wood burned at this facility at the time of sampling was small relative to other wood sources. Operators of facilities combusting substantial amounts of C&D debris wood should be mindful of potentially greater metal concentrations.

4.1.4 Comparison with Risk-Based Standards for Direct Exposure

To use WT ash beneficially where direct human contact with the ash is a plausible scenario, then comparison with risk-based standards is needed. Although risk assessment

though direct exposure is not the focus of this research, it is discussed briefly as it would certainly need to be considered as part of any beneficial use application. Total metal digestion was carried out on 30 samples. These samples include the 8 ash loads initially collected (each analyzed in triplicate) and the 6 individual WT ash samples subsequently collected (see chapter 4 for more details). It is worthy to note that the mean concentrations presented earlier (in Table 4-1) are only of the mixed sample concentrations. Thirteen metals were detected in the ash samples (aluminum, arsenic, barium, cadmium, cobalt, chromium, copper, vanadium, iron, lead, manganese, nickel and zinc). Since the data set is limited and consists of discrete samples, the US EPA suggests using a conservative estimate of the arithmetic mean concentration for each contaminant (EPA 2000). This estimate, the 95 percent upper confidence limit (UCL_{95}) of the mean is used to avoid underestimating the true mean. If the UCL_{95} on the mean for a contaminant is less than its risk-based guidance limit then the true mean of the contaminant should fall below that limit also (EPA 2001).

Table 4-2 compares the concentration for each metal with its respective Florida's risk-based soil clean up target levels (SCTLs) and the US EPA soil screening levels (SSLs). Florida's SCTLs examine direct exposure in two different scenarios, residential and industrial settings. Of the metals detected in WT ash, the UCL_{95} of only two metals (arsenic and iron) were above their respective Florida SCTL or the US EPA SSL. The iron UCL_{95} (35.6 g/kg) was above Florida's residential SCTL (23 g/kg) but below the industrial (480 g/kg). It is noted that iron does not have a US EPA SSL limit. Arsenic's UCL_{95} concentration in WT ash exceeded all of the risk-based standards presented. Unlike iron, arsenic is a carcinogen as well as a toxin. Based on the total metals analysis,

the primary limitation for the beneficial use of WT ash in a direct exposure scenario is the exceedance of the arsenic concentration.

Table 4-2: Total Metal Comparison with Risk-based Standards for Direct Exposure

Metal	Mean Unit Concentration ^a	UCL ₉₅ ^a	Florida SCTL		EPA SSL
			Residential	Industrial	
Al	3.59 ± 1.0	3.90	72	NA	NA
Fe	29.5 ± 18	35.6	23	480	NA
Zn	16.2 ± 3.5	17.4	23	560	23
As	35.8 ± 2.1	37.6	0.8	3.7	0.4
Ba	32.8 ± 6.0	35.3	5,200	87,000	5,500
Cd	2.20 ± 0.57	2.40	75	1,300	78
Cr	37.8 ± 10	41.4	210	420	390
Co	97.8 ± 36	110	4,700	110,000	NA
Cu	127 ± 57	147	110	76,000	NA
Pb	52.1 ± 13	56.6	400	920	400
Mn	260 ± 100	300	1,600	22,000	NA
Ni	13.2 ± 4.6	14.8	110	28,000	1,600
V	7.14 ± 2.2	7.88	510	7,400	550

^a Includes all the samples analyzed (8 initial loads, 1 mixed sample, samples over time)

4.2 Regulatory Batch Leaching Tests

The TCLP, WET, as well as a deionized water extraction test, were conducted on the composite WT ash sample in triplicate. The SPLP was carried out on the composite sample (in triplicate) and on the individual 6 samples collected over the eight-month period during 2001. Individual metal concentrations for the batch leaching tests are presented in Appendix G. These tests utilize different extraction solutions each with different initial pH values (as presented earlier) but do not specify a final extraction pH. As will be presented later, extraction pH is one of the leading factors controlling metals leachability. In WT ash, the buffering capacity and abundance of the calcium oxide resulted in a final extraction pH between 12.0 and 12.1 for the four batch tests. As

previously stated, a high buffering capacity at an alkaline pH is a characteristic of wood ash (Etiegni and Campbell 1991; Steenari et al. 1999).

4.2.1 Metal Leachability Using TCLP, WET and SPLP

Both the TCLP and the WET are used for hazardous waste characterization in US (WET is used specifically in California). If the concentration of one of the regulated metals in the leachate resulting from these tests is above its threshold limit (presented in Table 4-3) the solid waste is a toxicity characteristic (TC) hazardous waste. Metal concentrations in the TCLP and the WET leachates were an order of magnitude below the hazardous waste limits presented in RCRA and the California Code of Regulations (CCR). For example, the average TCLP and WET concentrations of lead in the composite sample composite samples were 0.055 and 0.12 mg/L respectively, compared to the RCRA and CCR hazardous waste regulatory limit of 5.0 mg/L. Previous research

Table 4-3: Mean Metal Concentrations in Batch Leaching Tests^a

Metal	Units	Detection Limit	TC Limit	TCLP Mean \pm SD	WET Mean \pm SD	SPLP Mean \pm SD	DI Mean \pm SD
Al	mg/L	0.007		< 0.007	1.32 \pm 0.1	< 0.007	< 0.007
Ca		0.1		1,490 \pm 50	2,970 \pm 170	1,070 \pm 53	628 \pm 30
K		0.1		167 \pm 7	392 \pm 9	153.3 \pm 5.7	165 \pm 8
Na		0.1		821 \pm 30	-	32.5 \pm 3.9	27.4 \pm 1
Zn		0.02	250 ^b	1.92 \pm 10	69.3 \pm 3	1.72 \pm 0.19	1.27 \pm 0.3
As	(μg/L)	5.0	5,000	< 5	102 \pm 1	< 5	< 5
Ba		40	100,000	207 \pm 10	149 \pm 60	218 \pm 80	95 \pm 10
Co		11	80,000 ^b	< 11	650 \pm 30	< 11	< 11
Cr		1.0	5,000	9.0 \pm 3	73 \pm 10	7.0 \pm 0.4	19 \pm 10
Cu		14	25,000 ^b	34 \pm 7	1,450 \pm 600	< 14	< 14
Fe		40		93 \pm 8	7,280 \pm 400	116 \pm 50	112 \pm 060
Mg		100		376 \pm 30	3,110 \pm 200	424 \pm 40	242 \pm 60
Mn		14		< 11	193 \pm 20	< 11	< 11
Ni		15	20,000 ^b	< 15	55 \pm 10	< 15	< 15
Pb		5.0	5,000	55.2 \pm 10	118 \pm 60	52.3 \pm 9	51.5 \pm 5

^a Averages reflect composite sample only

^b Toxicity limit applicable only to California's WET and is not a RCRA limit

suggested that citric acid (WET) complexes heavy metals to a greater extent relative to acetic acid (TCLP) (Bassi et al. 2000; Hopper et al. 1998). When compared with the TCLP, the WET extracted higher ($\alpha = 0.05$) concentrations of chromium, copper, iron, potassium, magnesium, and zinc. The WET detected nickel, manganese, arsenic and aluminum at measurable concentrations while the TCLP did not. However, for WT ash the WET and the TCLP extracted similar ($\alpha = 0.05$) concentrations of barium, calcium and lead. The high ionic strength of leachate may have been the cause of the similarity in the extracted concentrations. However, it is still unclear on the true nature of this similarity. Hooper et al. (1998) that lead leached from solid wastes to be greater in the WET when compared to the TCLP. However, Jang and Townsend found that the WET and TCLP extracted similar concentrations of lead from computer printed wire boards (Jang and Townsend 2003).

Unlike the TCLP, the SPLP simulates environmental conditions outside municipal landfills where organic acids resulting from degrading organic wastes are unlikely to be encountered. Studies conducted on MSW ash found that when compared to the TCLP, weakly buffered acid leaching solutions like the SPLP extract less metals (Cernuschi et al. 1990; Sawhney and Frink 1991). When metal concentrations extracted by the SPLP from WT ash are compared to those of the TCLP, copper was the only metal detected using the TCLP but not the SPLP. While the potassium concentration in the TCLP extraction solution was higher ($\alpha = 0.05$) than the SPLP solution, the TCLP extracted equal ($\alpha = 0.05$) concentrations of all other metals detected by both tests. Research has demonstrated the ability of the TCLP to extract more metals than the SPLP, in part because of the ability of acetic acid to complex heavy metals (Carey and Nagelski 1996;

Hopper et al. 1998). The hypothesized reason that this phenomenon was not observed in the WT ash tested was that the high ionic strength of the leachate (as a result of the readily leachable ions in the ash) reduces the chelating ability of acetate (Stumm and Morgan 1996). Reduction in the TCLP's complexation ability was also observed in other tests conducted on alkaline ash samples (Francis et al. 1984). The DI water extracted similar ($\alpha = 0.05$) metal concentrations as the SPLP.

Since SPLP was conducted on the initial composite sample and the subsequent samples collected over time, it was used as a tool to assess variability in metal leachability. The degree of variability in the SPLP batch leaching test results was lower than observed from the total metal digestion. This is attributed to the larger mass of WT ash used in the batch leaching tests (100-g) relative to that used in the total metals digestion (2-g). Over time, variability was also observed (Table 4-4). As explained

Table 4-4: SPLP Results for Ash Samples Collected Over Time

Metal	Metal Concentration (mg/L) ^a					
Date	1/24/2001	1/31/2001	2/8/2001	4/6/2001	4/12/2001	9/12/2001
Al	0.711	0.255	3.295	0.214	5.78	0.785
As	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005	< 0.005
Ba	0.116	0.242	0.313	0.299	0.251	0.143
Ca	1,420	1,070	1,050	904	1,930	1,039
Cd	< 0.0055	< 0.0055	< 0.0055	0.0062	0.1349	< 0.0055
Co	< 0.011	< 0.011	< 0.011	< 0.011	< 0.011	< 0.011
Cr	0.0240	0.0289	0.0117	0.0186	0.0385	0.0152
Cu	< 0.014	< 0.014	< 0.014	< 0.014	< 0.014	< 0.014
Fe	0.156	0.0716	0.0579	0.161	0.108	0.188
K	85.7	104	144	127	158	151
Mg	0.522	0.521	0.250	0.519	0.613	0.489
Mn	< 0.011	< 0.011	< 0.011	< 0.011	0.478	0.0275
Na	22.0	29.6	30.4	34.6	28.4	25.6
Ni	0.0305	< 0.015	0.0196	< 0.015	0.2451	0.0249
Pb	0.144	0.134	0.073	0.0801	0.0875	0.061
Zn	1.97	1.71	1.89	2.017	1.98	1.79

^a Concentrations represent analysis conducted on one sample

previously, differences may be related to changes in the combustion process, feedstock composition and operational changes over time (Bramryd and Fransman 1995; Buchholz and Landsberger 1995a; Glordano et al. 1983; Sawhney and Frink 1991).

4.2.2 USEPA Multiple Extraction Procedure

The MEP is used to assess long-term leaching from solid wastes. The procedure involves sequential extractions of the same waste sample. In most cases, leachable metal concentrations are expected to decrease with each subsequent extraction as less contaminant is available for leaching. The MEP was conducted in triplicate on the WT ash, and of the 17 metals analyzed, arsenic, cobalt, cadmium, copper, manganese, and nickel were below their respective detection limits (see Table 4-3). Because the concentrations of aluminum and iron were close to their detection limits (7 and 20 $\mu\text{g/L}$ respectively), the results were somewhat variable, and therefore these two metals were not examined further. Sodium concentrations were not reported because sodium acetate was used in the first extraction step.

Initially, the calcium (Figure 4-1), chromium (Figure 4-1), barium (Figure 4-2), potassium (Figure 4-3), and zinc (Figure 4-3) concentrations decreased with consecutive extractions as expected (see Appendix H). As the test proceeded and fresh extraction fluid was added to the same WT ash, the pH dropped, as less leachable alkalinity was available to neutralize the acid added. After the initial decrease, the metal concentrations stabilized and in some cases began to increase. One potential cause of the increased leaching is the deterioration of the outer layer of the ash particle. Poon and Chen (1999) reported that the removal of calcium and sodium from the surface layer of an ash particle might cause the surface matrix to degrade releasing trapped metals. Another probable

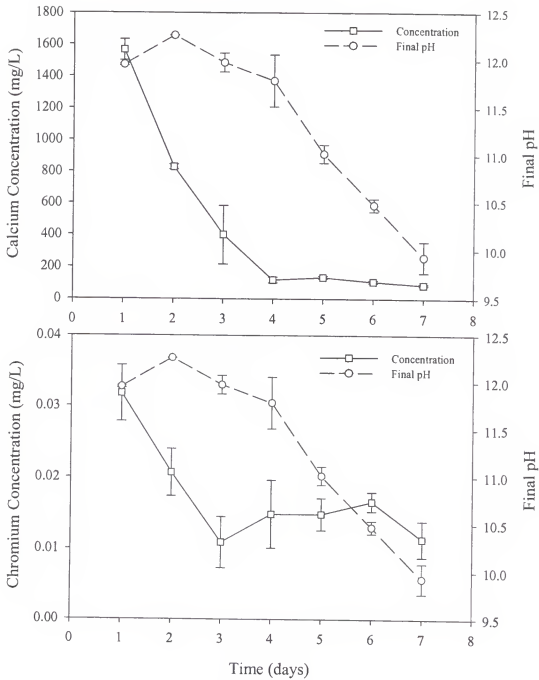


Figure 4-1: Calcium and Chromium Leachability in US EPA's MEP

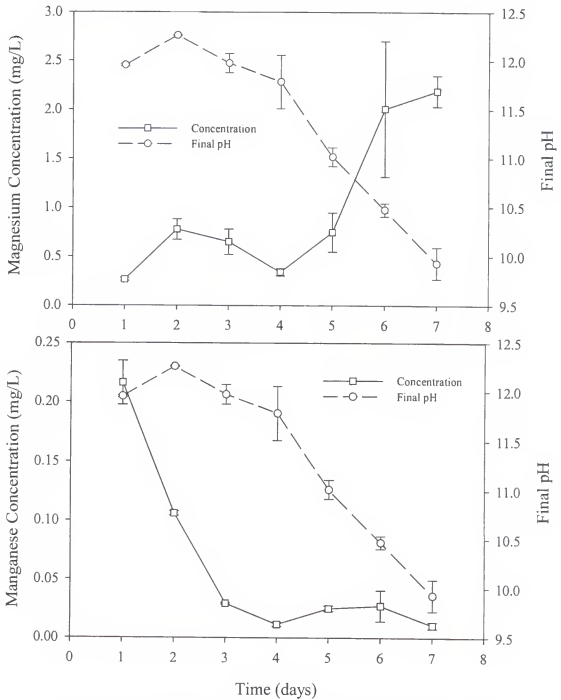


Figure 4-2: Magnesium and Barium Leachability in US EPA's MEP

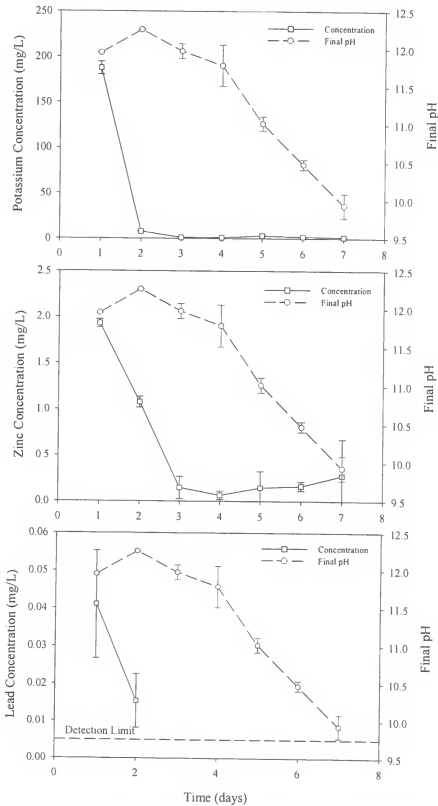


Figure 4-3: Potassium, Zinc and Lead Leachability in US EPA's MEP

cause is the impact of changing pH on metal leachability. The effect of changing pH is most apparent in the magnesium leachability trend. Magnesium concentrations increased throughout the test (Figure 4-2). As will be presented in the next section, when the pH of the leaching solution drops, the leaching of magnesium increases.

While the increased leaching of magnesium over time might not be the cause of great environmental concern, this phenomenon bears further discussion because of the potential impact of pH changes on the leachability of more toxic metals. For example, lead was only detected in the first two extractions, and arsenic was not detected at all. In theory, if the pH continued to drop with additional extractions beyond the seven conducted here, the leachability of both of these metals could increase (see figures 4-5 and 4-9 discussed in the next section). On the other hand, one must also factor in the time period associated with the test. The developers of the MEP proposed that each extraction step represented approximately 100 years. Under a more conservative scenario (760 mm/yr (30 in/yr) of rainfall infiltrating into a layer of ash 0.9 m (3ft) in thickness), seven extraction steps of the MEP would represent 168 years of infiltration. If the goal of conducting a risk assessment was to evaluate possible risks this far into the future, continued leaching extraction steps until the pH stabilized might be warranted.

4.3 Factors Affecting Metals Leachability

4.3.1 Extraction pH

Research examining leachability of metals from solids has reported that pH is the major controlling factor in determining mobility (Fytianos et al. 1998; Kanungo and Mohapatra 2000; Li et al. 2001; Van der Sloot 2002; Van der Sloot et al. 2001). To examine the effect of extraction solution pH on metal leachability from WT ash, batch leaching tests were carried out at different leaching solution pH values ranging from 1 to

13. All 17 metals analyzed were detected at pH values less than 4. But as pH values increased above 4, chromium, cadmium, iron and vanadium concentrations fell below their respective detection limits. Figures 4-4 through 4-9 present metal concentrations in leachates (mg/L) as a function of extraction pH. Metals concentration at a given pH is tabulated in Appendix J.

pH had varying effects on metal solubility. The solubility of some metals increased under extreme pH values while for others the change in pH had no effect (Chuan et al. 1996; Van der Sloot et al. 1997). The concentrations of sodium, potassium (Figure 4-4) and calcium (calcium not presented) were not affected by the change in pH. Campbell (1990) suggested that solubility rather than pH controls potassium, calcium and sodium availability from wood ash. Lead solubility (Figure 4-5) was highest under extreme pH values ($\text{pH} < 2$ and $\text{pH} > 12$) and lowest around pH 9. The increase in metal solubility under alkaline conditions is attributed to the formation of soluble metal hydroxides under extreme alkaline conditions (Fytianos et al. 1998).

This behavior was also observed with aluminum (Figure 4-5), barium (Figure 4-6), copper (Figure 4-6), cobalt (Figure 4-7), nickel (Figure 4-7), manganese (Figure 4-8) and zinc (Figure 4-9). Other researchers also observed similar leaching trends for a variety of solid wastes (Cernuschi et al. 1990; Fallman 2000; Kjledsen and Christensen 1990; Kosson et al. 1996; Van der Sloot 2002; Van der Sloot et al. 1997; Van der Sloot et al. 2001). Steenari et al. (1999) concluded that under acidic conditions ($\text{pH} < 4$) dissolution of wood ash increased significantly thus the solubility of metals also increased. This phenomenon was also observed in WT ash. At pH values less than 4, metal leachability

was high relative to that at other pHs. However, in this pH range (pH less than 4) the change in pH did not greatly affect the solubility of these metals.

Magnesium was extracted at constant concentrations until approximately a pH of 6, where it dropped dramatically, as presented in Figure 4-8. Although arsenic (Figure 4-9) was relatively leachable under acidic conditions, it demonstrated a large decrease in concentration between pH 1 and 5. The rate of arsenic decrease leveled off at pH values above 6. At pH values above 10, arsenic was not detected as the concentration fell below the instrument's detection limit of 0.07 mg/kg. The change in metal release as a function of pH has a great implication on management options for the ash. If the environmental pH of the management option for the ash is acidic (MSW landfill), then metals (like arsenic) that are not soluble under neutral or alkaline conditions will solubilize. Thus care must be taken when management options are examined.

4.3.2 Contact Time Effects

One of the main reasons to examine metal leachability as a function of time is to determine whether the time requirements for most batch tests (18 hours) are sufficient to achieve chemical equilibrium. Contact time effects on metal leachability were investigated by placing a given mass of WT ash in contact with SPLP extraction fluid for various periods of time. In the same fashion as presented earlier for the SPLP, 9 of the 17 metals analyzed were detected above their detection limits. Metal concentrations (mg/L) were plotted as a function of extraction time as presented in Figures 4-10 through 4-12. These concentrations are tabulated in Appendix K. As expected, metal concentration increased with time until reaching steady state. A convenient method to express metal concentration as a function of time is through mathematical modeling. Mathematical

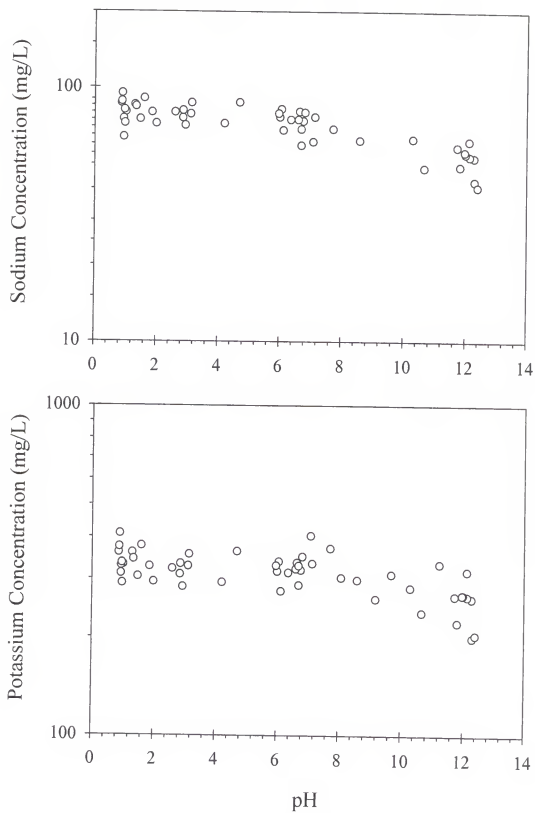


Figure 4-4: Sodium and Potassium Leachability as a Function of pH

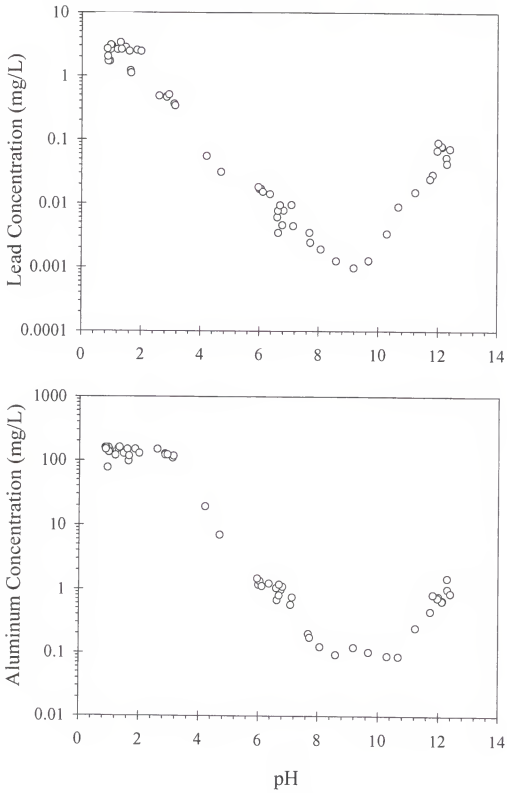


Figure 4-5: Lead and Aluminum Leachability as a Function of pH

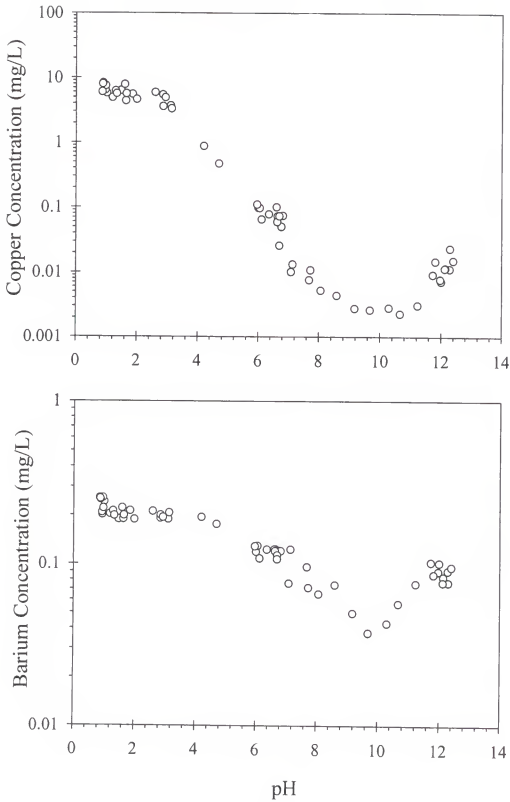


Figure 4-6: Copper and Barium Leachability as a Function of pH

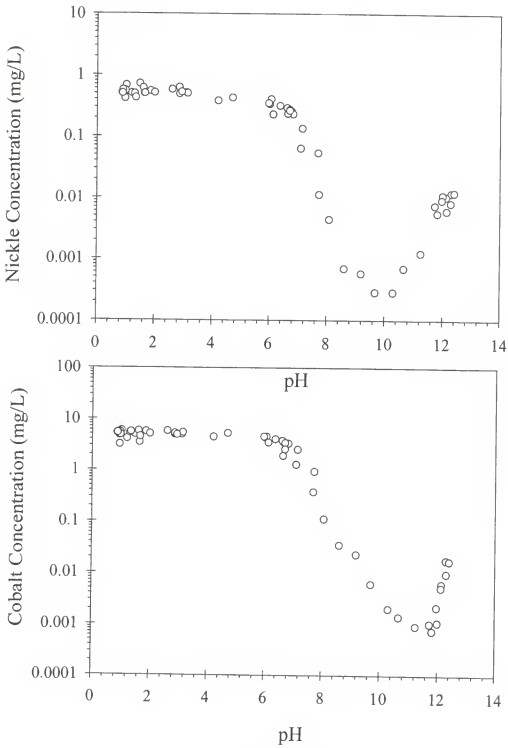


Figure 4-7: Nickel and Cobalt Leachability as a Function of pH

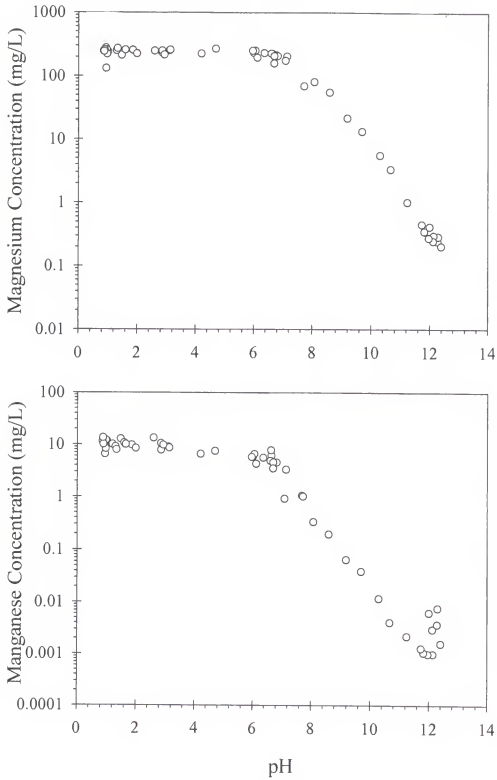


Figure 4-8: Magnesium and Manganese Leachability as a Function of pH

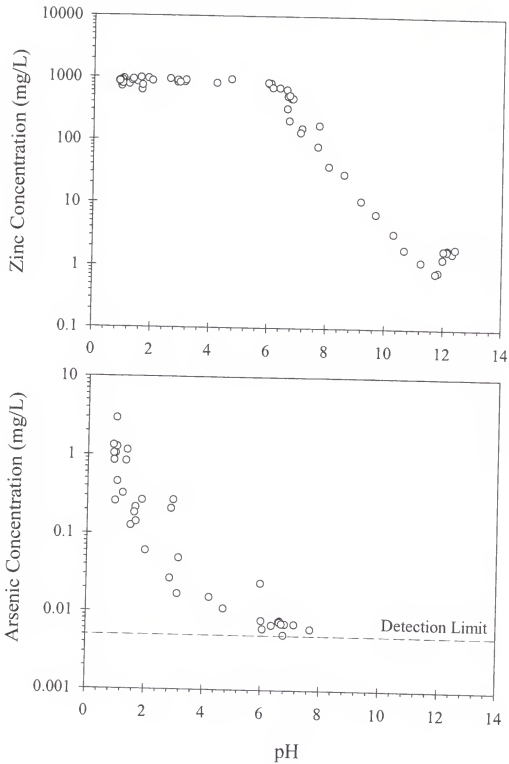


Figure 4-9: Zinc and Arsenic Leachability as a Function of pH

models were developed to better evaluate the effect of contact time on metal leachability (Janos et al. 2002; Van Herck et al. 2000). The main chemical reactions occurring during extractions may be mathematically modeled using equation 4-1 and are presented as solid lines in the figures.

$$C_t = C_0(1 - e^{-kt}) \quad 4-1$$

C_t is the metal concentration in a solution at time (t), C_0 is the metal concentration at $t = 0$, and k is a coefficient. C_T , k , and R^2 values for the 9 metals are tabulated in Table 4-4. The t_{95} , where 95% of the available metal is leached, was also calculated and presented in Table 4-5. The t_{95} of more than half of the metals detected was less than 3 hours, and none exceeded 18 hours. A two-year study concluded that the release of zinc and iron from wood ash is diffusion controlled and a relatively quick process (Steenari et al. 1998). Steenari's results agreed with other literature investigating alkaline ashes with

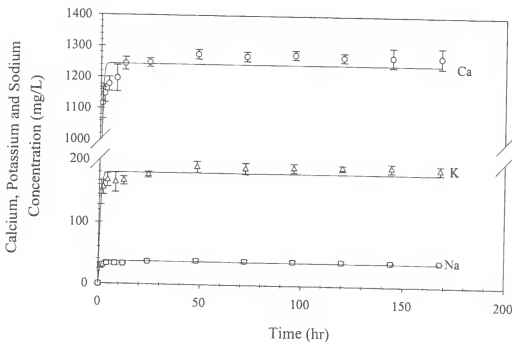


Figure 4-10: Calcium, Potassium and Sodium Leachability as a Function of Time

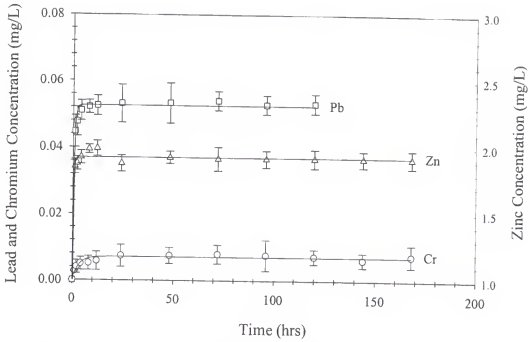


Figure 4-11: Lead, Zinc and Chromium Leachability as a Function of Time

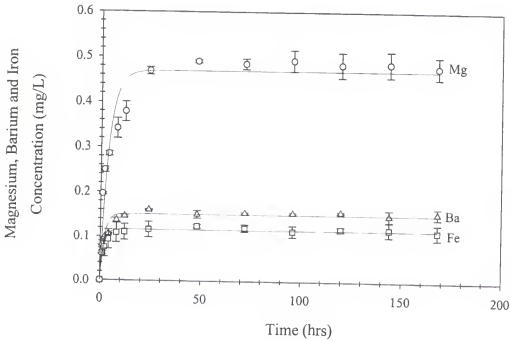


Figure 4-12: Magnesium, Barium and Iron Leachability as a Function of Time

respect to the time to reach equilibrium (Buchholz and Landsberger 1995a; Fytianos et al. 1998; Kanungo and Mohapatra 2000). Thus, the time required for the three regulatory batch tests examined earlier (TCLP, WET, and SPLP) was sufficient to achieve steady state concentrations. The longer extraction time used for the WET (48 hours) relative to the TCLP (18 hours) should not have played a role in the statistically higher metal concentrations extracted by the WET over the TCLP.

Table 4-5: Time Study Results

Metal	C₀ (mg/l)	-k (h⁻¹)	R²	t₉₅ (hrs)
Na	36.7	1.32	0.944	2.3
Ca	1243	2.15	0.988	1.4
K	180	1.50	0.956	2.0
Ba	0.151	0.501	0.944	6.0
Fe	0.116	0.565	0.969	5.3
Mg	0.469	0.263	0.918	11.4
Zn	1.92	3.28	0.996	1.0
Pb	0.0525	2.68	0.993	1.1
Cr	0.00697	0.314	0.904	9.5

4.3.3 Liquid-to-Solid Ratio Effect

It is unclear if contaminant concentrations in batch leaching tests (like the SPLP) represent the pore water concentrations in the waste as it might be applied in the environment, or whether they represent diluted concentrations that might be expected in the groundwater (this point will be discussed in details in chapter 7). Because of the relatively high liquid-to-solid ratios associated with batch tests (20:1, 10:1), some state guidelines (e.g., Florida and Minnesota) assume that dilution occurs during the test, and that the results should be compared directly to the appropriate water quality standards (DEP 2001). On the other hand, the EPA contends that the concentration in batch test leachates represent those encountered in the pore volume of the waste, and thus an

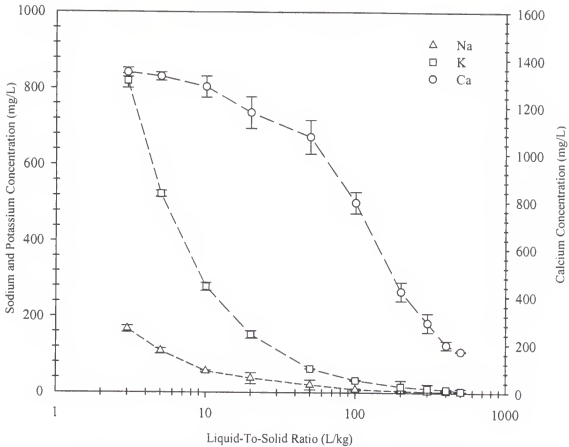


Figure 4-13: Sodium, Potassium and Calcium Leachability as a Function of Liquid-to-Solid Ratio

appropriate dilution factor should be applied to the batch test results (EPA 2001) (see chapter 2 for more discussion). Figures 4-13 through 4-15 present the results of the experiments performed to examine the impact of liquid-to-solid ratio. Actual concentrations are tabulated in Appendix L. As observed by other researchers, results varied between metals (Bordas and Bourg 2001; Fytianos et al. 1998; Kanungo and Mohapatra 2000). The concentrations of both potassium and sodium (Figure 4-13) showed a sharp decrease with increasing liquid-to-solid ratio. The rate of the decrease in the concentration of calcium (Figure 4-13), magnesium, iron and barium (Figure 4-14), however, was lower than that of potassium and sodium. Thus the difference in

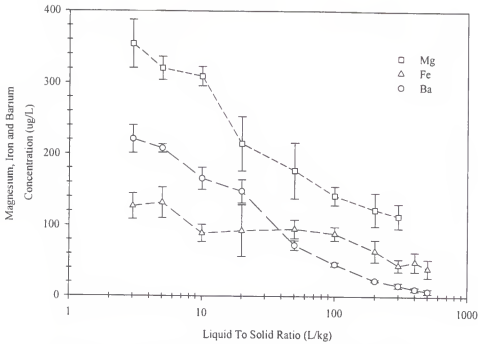


Figure 4-14: Magnesium, Iron and Barium Leachability as a Function of Liquid-to-Solid Ratio

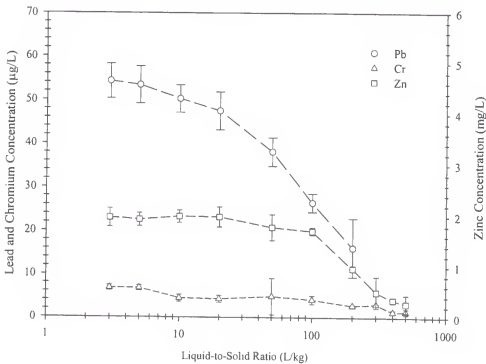


Figure 4-15: Lead, Chromium and Zinc Leachability as a Function of Liquid-to-Solid Ratio

concentration of these metals in batch tests, as liquid-to-solid ratio increases, is lower. Lead, chromium and zinc (Figure 4-15) concentrations remain relatively constant (52 $\mu\text{g/L}$, 7 $\mu\text{g/L}$ and 1.8 mg/L respectively) up to liquid-to-solid ratios of 20:1, 300:1 and 100:1 respectively. Thus for these three metals the results of batch leaching test such as the SPLP indicate pore water concentration. The results of these batch test demonstrate the uncertainty associate with applying a batch leaching test results to estimate groundwater contamination (see chapter 2.4). For example, the lead concentration at a liquid-to-solid ratio of 20:1 is indicative of pore water concentrations (concentration is similar to that at a lower liquid-to-solid ratios). However, at the same liquid-to-solid ratio (20:1) the potassium concentration is diluted relative to lower liquid-to-solid ratios. This point will be further discussed in chapter 7.

4.4 Characterization Summary

The major conclusions of this chapter were summarized and presented as follows:

1. Ash from the combustion of waste wood and scrap vehicle tires was assessed.
2. All 17 of the metals analyzed were detected in WT ash samples.
3. Relative to concentrations of metals in ash from the combustion of wood presented in the literature, the WT ash was elevated in concentrations of iron, zinc, copper, cobalt and sodium.
4. The WT ash had similar concentration of lead as that found in wood ash alone presented in the literature.
5. When total metal concentration of WT ash was compared to Florida SCTLs for direct exposure, the arsenic concentration exceeded its residential as well as industrial SCTL. The iron concentration exceeded its residential limit.
6. Of the regulatory batch leach tests performed, in most cases the WET extracted greater concentrations of metals than the TCLP. That was caused the ability of citric acid (WET) to chelate metals more the acetic acid (TCLP).

7. The TCLP extracted similar concentrations as those measured using the SPLP and deionized water. This was caused by the high ionic strength that inhibited the effectiveness of the acetic acid.
8. Results of both the TCLP and the WET showed that the WT ash tested was not a characteristic hazardous waste.
9. pH was found to be a major factor controlling the concentrations of metals leached, with all metals being detected in leachates at low pH values (less than 4). Minimum leachability was typically observed in the pH range of 9 through 11. Some metals showed a concentration increase at pH values greater than 11.
10. Experiments examining contact time found that the batch test time periods were sufficient to reach equilibrium.
11. The study of liquid-to-solid ratio impact suggested that concentrations of metals such as lead and zinc in the batch test leachates were representative of pore water concentrations.

CHAPTER 5

LYSIMETER RESULTS

This chapter presents the results of the lysimeter studies. The leachate generation data are presented first followed by a discussion of the conventional parameters. Next, anions, cations and metal concentrations in the leachate are presented. This chapter also presents a statistical comparison between the different lysimeter types. The control column concentrations were consistently below detection limit. Thus they were not included in any of the figures. As mentioned earlier (chapter 3), type 1 lysimeters contain 0.3 m (1 ft), type 2 lysimeters contain 0.6 m (2 ft) and type 3 lysimeters contain 0.9 (3 ft) of WT ash. Each lysimeter type consisted of three triplicate lysimeters (see Figure 3-1). Thus, at a given volume, the mean of the triplicate lysimeters leachate contaminant concentration was calculated and plotted as a function of the cumulative volume drained (V_L). V_L was calculated using equation 5-1. Standard deviation was not included because it made reading the graphs rather difficult.

$$V_L = \sum_{i=0}^n V_i \quad (5-1)$$

where V_L is the cumulative volume, t is the draining event, n is the number of drain events, and V_i is the volume of leachate collected at each drain event. The student t-Test (presented in chapter 3) was used as a statistical method to compare leachate quality parameters as a function of lysimeter type. In an effort to eliminate the effect of the initial-wash off phase on the statistical analysis, only concentrations of contaminant

corresponding to a cumulative volume of 10 L and higher were used in the analysis. Raw lysimeter data are tabulated in Appendix M.

5.1 Leachate Generation

Approximately 70 L (18.5 gal) of synthetic precipitation leaching procedure (SPLP) leaching solution were added to each lysimeter at the rate of 5ml every 30 min. The volume of SPLP solution added to each lysimeter corresponds to approximately 384 cm (151 inch) of rainfall or almost 3 years of rain fall in Florida (average rain fall 52 inch per year). Since the ash depth in the lysimeters was varied depending on the lysimeter type (0.3, 0.6, or 0.9 m), the time required for the initial leachate production was somewhat variable between lysimeters. It required type 1 lysimeters six weeks of leachate addition before leachate was generated. Type 2 lysimeters lagged one week and type 3 lysimeters two weeks behind type 1 lysimeters. Leachate generations from the lysimeters averaged 60L per lysimeter.

5.2 Conventional Parameters

The pH values of the lysimeter leachates varied between 11.5 and 13 for all three lysimeter types as presented in Figure 5-1. Previous characterization of WT ash detected a high concentration of calcium oxide (Chapter 4). Upon hydration and disassociation in water, calcium oxide releases hydroxide molecules. The addition of these molecules helps in maintaining an alkaline pH (Etiegni and Campbell 1991). The alkaline leachate pH was coupled with hydroxide alkalinity (Figure 5-1), which is prevalent in wood ash incinerated at temperatures above 1000 °C (Etiegni and Campbell 1991). The leachate ORP measurement is a useful index of the oxidative state of the leachate. Positive ORP measurements indicate an oxidative state while negative ones indicate a reduced state (Snoeyink and Jenkins 1980). Because of the difficulty of interpreting ORP, these

measurements need to be used for qualitative purposes (Stumm and Morgan 1996). The leachate ORP ranged between -150 and 50 mV as presented in Figure 5-1. The ORP increased during the initial stages of the experiment, it started decreasing in all three lysimeter types during the later part of the experiment. Swelling of the ash when hydrated (Etiegni et al. 1991) may have reduced the migration of oxygen into the columns lowering the ORP (Figure 5-1). The high concentration of iron in the ash (see chapter 4) may also play a role in the lowering of the ORP. When iron is hydrated, it is reduced and lowers the ORP even further (Meng et al. 2001; Stumm and Morgan 1996).

Electric conductivity, which is a measure of the ability of a solution to conduct a current and is attributed to the ions present, of the lysimeter leachate was initially high. After the initial wash-off phase, the conductivity of all three lysimeter types reached a relatively constant value around 10 ms/cm. Research has demonstrated that ash leaches rapidly at first (wash-off phase), and then approaches steady state leachate concentrations (Edit et al. 1992; Farquhar 1989; Lu 1996). Mudd and Kodikara (2000) concluded that the initial wash off phase represents the rapid advective leaching of highly soluble surface salts with subsequent decrease in concentrations. On the other hand, the steady state leachate concentrations are more controlled by diffusive leaching fluxes within the ash matrix. The total dissolved solids (TDS) followed the same trend as conductivity, reaching a steady value around 5 mg/L. The none-purgable organic carbon (NPOC) as well as chemical oxygen demand (COD) as presented in Figure 5-3 also followed a similar leaching trend. NPOC is mainly attributed to the residual carbon left in the ash after incineration while COD is controlled by the inorganic oxidative contaminants like iron and chromium.

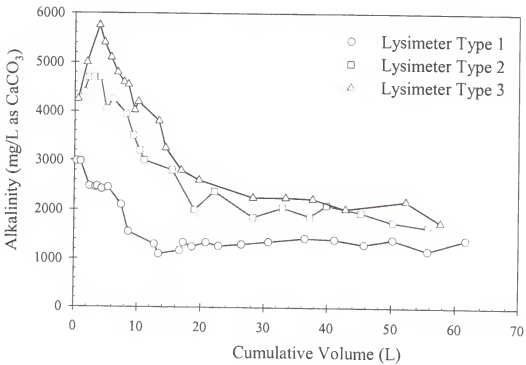
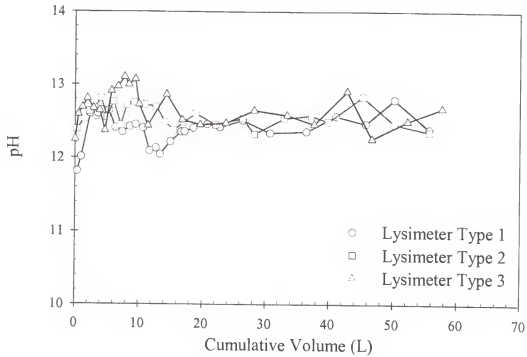


Figure 5-1: Lysimeter pH and Alkalinity as a Function of Leachate Cumulative Volume

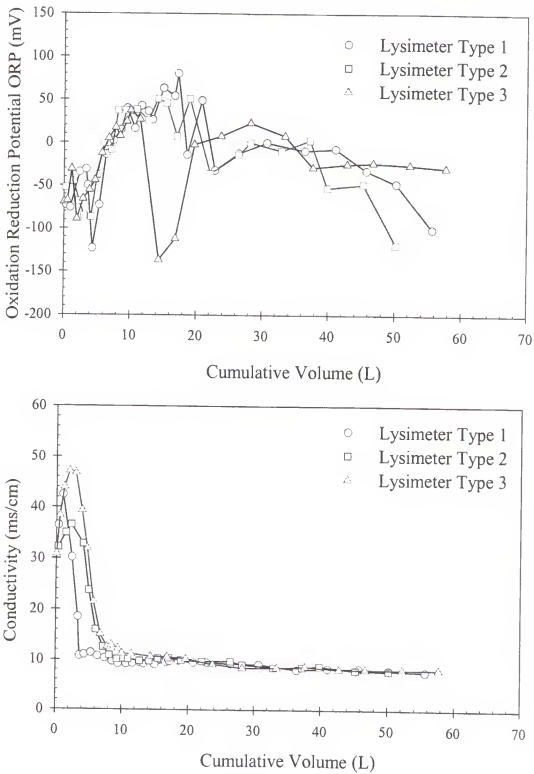


Figure 5-2: Lysimeter ORP and Conductivity as a Function of Leachate Cumulative Volume

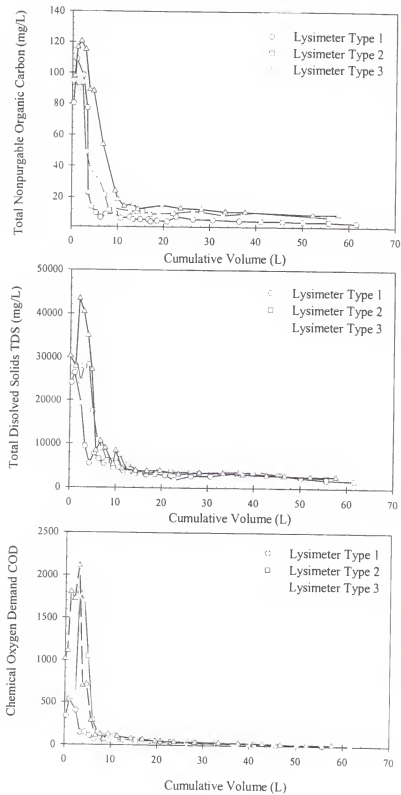


Figure 5-3: Lysimeter TOC, TDS and COD as a Function of Leachate Cumulative Volume

While the depth of the ash layer statistically (using student T-test, presented in chapter 3, with $\alpha = 0.05$) did not have an effect on the conventional parameters measured except for alkalinity. Although alkalinity was statistically different between the three lysimeter types, pH does not change. A unit change in pH corresponds to 5000 mg/L change in alkalinity. Thus small variability in lysimeter type pH will contribute to large variability in alkalinity. The reason alkalinity is different between the lysimeter types is because of the different mass of ash used in the different lysimeter types, the available hydroxide in lysimeters type 1 is less than type 2 or 3. Thus the leached concentrations reflect that difference. Other parameters however, showed differences mainly in the initial wash-off phase. There was an apparent delay in the initial phase (lysimeter type 3 lagged behind 2 which in turn lagged behind 1).

5.3 Anions

Three anions (chloride, bromide and sulfate) were consistently detected in the lysimeter leachate as presented in Figure 5-4. The maximum chloride concentrations of 55 g/L (lysimeter type 1) and 160 g/L (both lysimeter types 2 and 3) were achieved during the wash-off phase. The chloride concentrations in the leachate decreased quickly until reaching a steady state concentration (approximately 5 mg/L) at V_L values greater than 10 L. Relative to chloride, bromide leached at lower concentrations. The maximum bromide concentrations (10, 11 and 20 g/L for lysimeter types 1, 2 and 3 respectively) were achieved during the initial phase of the experiment. The bromide concentration also reached a steady state (4 mg/L) at V_L values around 10 L. Sulfate leached in the same manner as bromide and chloride. However, its steady state concentration was noticeably higher. Sulfate reached a steady state concentration around 100 mg/L. Although wood

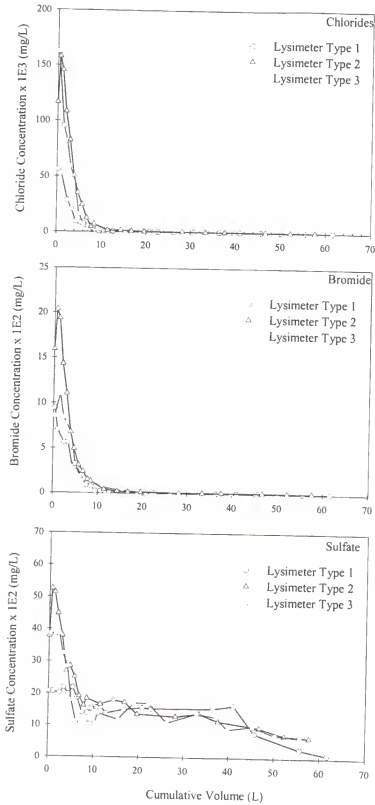


Figure 5-4: Ash Leachate Anion Concentrations as a Function of Cumulative Volume

ash contains low concentration of sulfate (1% by mass) (Misra et al. 1993), tires contain high concentration of sulfur (approximately 15% by mass) (Levie et al. 1995), which contributes to the steady state concentrations relative to the other anions.

Statistically ($\alpha = 0.05$), the anion concentrations were similar in all lysimeter types regardless of depth. As presented in Figure 5-4, the difference in anion concentrations between the three lysimeter types was only apparent in the early wash-off phase of the experiment, after which the three lysimeter types leached relatively equal anion concentrations. Steady state anions concentrations are controlled by diffusive leaching fluxes within the ash matrix. Diffusive leaching is a slow process (Mudd and Kodikara 2000).

5.4 Cations

Four cations (sodium, potassium, magnesium and calcium) with two distinct leaching trends were detected in the leachate. The first leaching trend, similar to that of anions, was observed in the leaching of sodium, potassium and magnesium. The concentration of these cations was initially high then decreased quickly to reach a steady concentration as presented in Figures 5-5 and 5-6. Type 3 lysimeter leachate contained the highest concentrations of sodium and potassium (2,000 and 10,000 respectively). The sodium and potassium concentrations rapidly decreased to reach a steady state concentration (less than 200 mg/L for both sodium and potassium) at V_L values greater than 10 L for all three lysimeter type. The maximum leachable concentrations of potassium and sodium were similar to those observed in wood ash. Etiegni and Campbell (1991) examined the leachability of potassium and sodium from wood ash and concluded

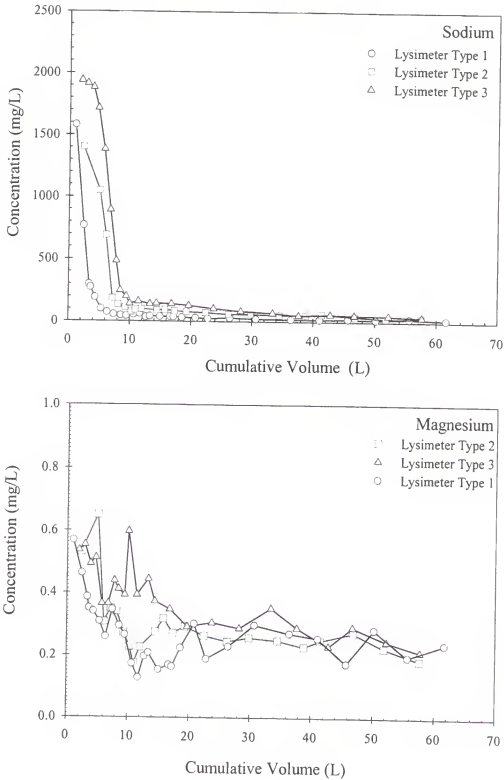


Figure 5-5: Lysimeter Sodium and Magnesium Concentration as a Function of Cumulative Volume

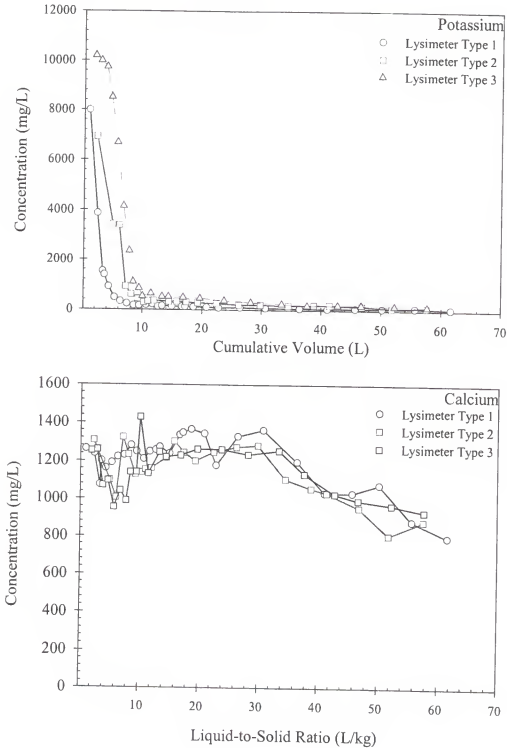


Figure 5-6: Lysimeter Potassium and Calcium Concentration as a Function of Cumulative Volume

that the maximum extractable concentration to approximately be 11,000 and 3,000 mg/L for potassium and sodium respectively. The concentrations of potassium and sodium leached from WT ash, however, were different than coal ash. A large-scale research conducted on coal ash concluded that the maximum extractable concentration of potassium and sodium were 130 and 5,800 respectively (Mudd and Kodikara 2000). These differences are attributed to the different fuel stream being used.

Of the cations the leached magnesium concentration was the lowest. The maximum magnesium concentration of 0.7 mg/L (type 2 lysimeters) was detected early on in the experiment then concentrations decreased to approximately 0.2mg/L. Magnesium concentration leached from wood ash alone was also observed to be the lowest of all cations. The maximum magnesium concentration leached from wood ash was 0.05 mg/L (Etiegni and Campbell 1991). The low leachability of magnesium might be explained by the extraction pH. The pH static experiment (see chapter 4) demonstrated that magnesium solubility at alkaline pH (like the lysimeters leachate) is low. Thus the concentration of magnesium was the least of all cations present in the ash.

A second trend was observed in the leaching of calcium (Figure 5-6). The calcium concentrations in the leachate were relatively constant (1300 mg/L) throughout most of the experiment. Because of the high calcium concentration in the ash (see chapter 2), the leaching of calcium was solubility controlled (CaO solubility in water is 1310 mg/L) rather than diffusion controlled. The calcium concentration, however, showed a slight decrease at V_L values greater than 50 L.

Statistically, the depth of the ash layer did not have an effect on the concentration of cations in the leachate ($\alpha = 0.05$). Differences in leached cation concentrations were

only visible early in the experiment (Figures 5-5 and 5-6). Therefore no statistical differences are present between the three lysimeter types. This similarity can be explained in the same manner as similarity with the anion concentrations.

5.5 Metals

Five metals (aluminum, iron, barium, lead and zinc) were detected consistently in the leachate. Arsenic, chromium, nickel, and selenium were not detected above the instrument's detection limit. The aluminum, barium and iron concentrations showed a general decreasing trend with increasing V_L as presented in Figure 5-7 and 5-8. At the beginning of the experiment, the aluminum concentration was briefly high in lysimeter type 1. However, the aluminum concentration decreased quickly to steady state around 0.2 mg/L. The iron concentration in all three lysimeter types reached a steady state around 80 $\mu\text{g/L}$ at V_L values greater than 10 L. The decrease in iron concentration may be attributed to the decrease in ORP. As the ORP decreases, indicating a reduced environment, iron is reduced from Fe^{+3} to the less mobile Fe^{+2} . The leached barium did not appear to achieve steady state and its concentrations in all three lysimeter types did not level off even at relatively large V_L values. Unlike the anions and the cations, the wash-off phase was not as well defined. This may indicate that most of the metals are trapped within the ash particles, thus most of the leaching occurring is diffusion control.

The leached zinc concentrations for all three lysimeter types were relatively steadily varying between 1.5 and 2.0 mg/L (Figure 5-9). The relatively steady leaching trend associated with zinc indicates that zinc's leachability could have been solubility controlled. Most of the zinc found in the lysimeter leachate was possibly a result of the

addition of tires to the fuel stream. Tire bottom ash contains approximately 52% zinc (Levie et al. 1995). Zinc was not detected in the leachate of lysimeters examining wood

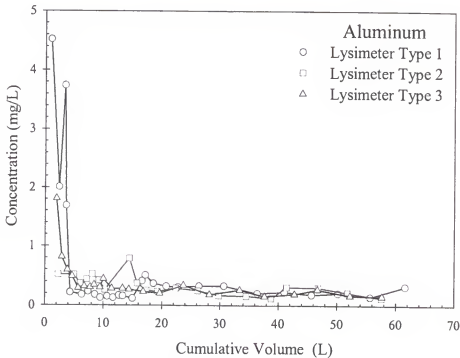


Figure 5-7: Lysimeter Aluminum Concentration as a Function of Cumulative Volume ash leachability (Xiao et al. 1999). Zinc leachability from coal ash was also lower than WT ash. A lysimeter study examining the leachability of coal ash concluded that the maximum zinc concentration from coal ash was 0.55 mg/L (Hjelmar 1990).

The leached lead concentration increased, as presented in Figure 5-9, with increasing cumulative volume as presented in Figure 3-9 until V_L of 10 L, after which the lead concentrations reached a steady value of approximately 55 $\mu\text{g/L}$. The initial low lead concentration may have been caused by the lead adsorbing to the gravel in the drainage layer and may be also the walls of the tedlar® storage bags (see chapter 3 for lysimeters set up). As the lead leaches pass through the gravel it may sorbs onto the gravel, as a result the soluble lead concentration in the leachate may be lowered.

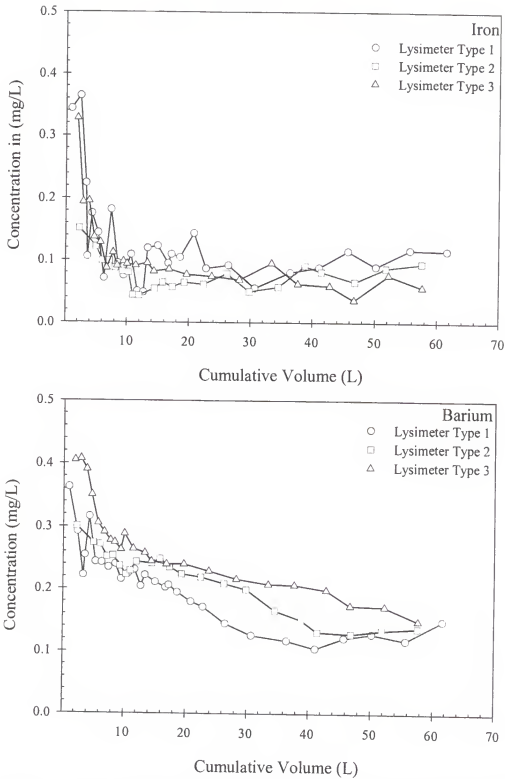


Figure 5-8: Lysimeter Iron and Barium Concentrations as a Function of Cumulative Volume

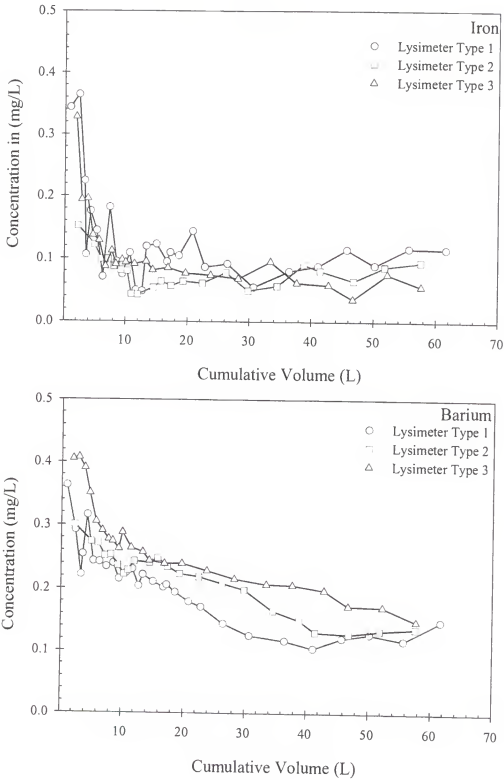


Figure 5-8: Lysimeter Iron and Barium Concentrations as a Function of Cumulative Volume

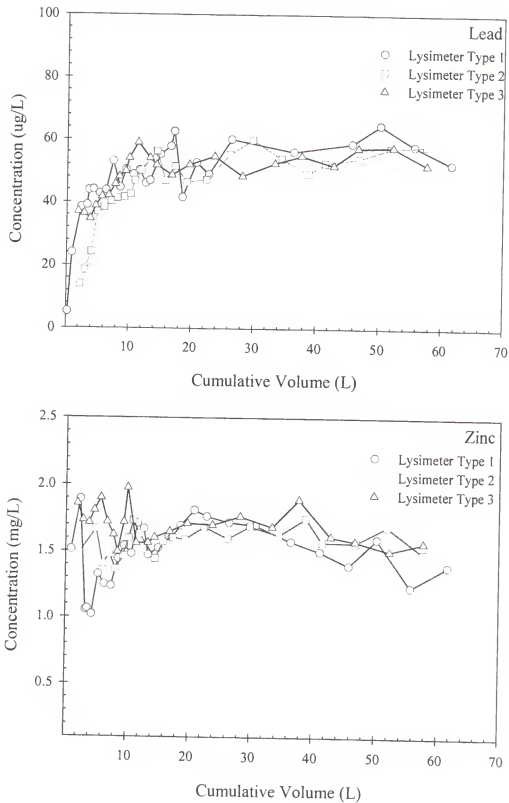


Figure 5-9: Lysimeter Lead and Zinc Concentration as a Function of Cumulative Volume

However, as these sorption sites get filled, less lead may sorb and more of the lead could remain in lysimeter leachate thus increasing its concentration. Shendriakar et al. (1976) found that for pH values above 3.5, the adsorption of lead on the walls of containers ranged from 10 to 20%. The adsorption of lead has also been found to increase with increasing pH (Scrudato and Estes 1975). The steady state concentration of lead was similar to that of wood ash alone (Xiao et al. 1999), but less than that present in some coal ash (Hjelmar 1990).

Statistically ($\alpha = 0.05$), the depth of ash did not have an effect on metal concentration in the leachate. As stated earlier, the leachability of metals present in the ash was mainly controlled by diffusive leaching fluxes within the ash matrix (Mudd and Kodikara 2000).

In general the leachate quality is similar to that of wood ash alone. The pH of the WT ash lysimeter is similar to that present in wood ash alone (Steenari et al. 1998; Steenari and Lindquist 1999). The leachability of anions and cations is also similar to that of wood ash alone (Steenari et al. 1998). With respect to metals, most metals leached from WT ash leached at similar concentration as wood ash alone. Lead leachability, in particular, raises concern over the amount of lead that might leach from wood ash alone. The leached lead concentration (steady state concentration 53 $\mu\text{g/L}$) is above drinking water standard. Potential groundwater contamination from the land application of wood ash alone should be better examined. Zinc leached at a higher concentration than wood ash alone (Xiao et al. 1999). This probably was caused by the

addition of tires to the fuel stream. Tires contain large concentration of zinc (18.2 g/kg).

Over all, the leaching behavior of WT ash is similar to that of wood ash alone.

5.6 Summary of Lysimeter Leaching

- 1 Three different leaching trends, presented graphically in Figure 5-10, were observed during the leaching of wood and tire ash:
 - a. Steady concentration of a particular contaminant throughout the leaching process
 - b. A rise in concentration of a contaminant until reaching a maximum steady state concentration
 - c. A decay in the concentration as the experiment progressed until reaching a minimum concentration.

Most of the parameters examined followed these leaching trends and a summary of the parameters; leaching trend (a, b or c) and steady state concentration are presented in Table 5-1.

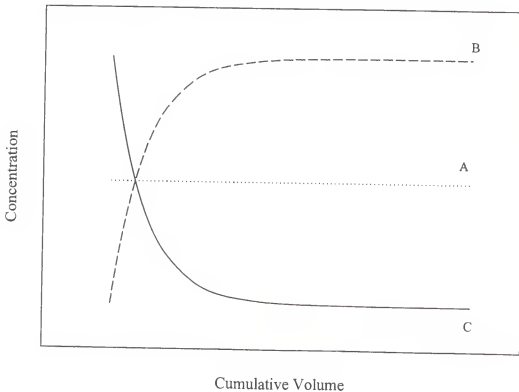


Figure 5-10: Schematic of Different Leaching Trends

Table 5-1: Summary of Lysimeter Leaching Trends and Steady State Values

Parameter	Leaching Trend	
	Type	Steady State Value
PH	A	12.5
ORP	A	-50 mV
Calcium	A	1300 mg/L
Zinc	A	1.7 mg/L
Lead	B	0.055 mg/L
Alkalinity	C	2000 mg/L as CaCO ₃
Conductivity	C	10 ms/cm
COD	C	5
TDS	C	300 mg/L
NPOC	C	10 mg/L
Potassium	C	100 mg/L
Chloride	C	10 mg/L
Sulfate	C	200 mg/L
Bromide	C	5 mg/L
Sodium	C	30 mg/L
Magnesium	C	0.02 mg/L

2. The leachate characteristics of the lysimeters with different ash depth were initially different which was caused by the initial wash off phase. However, steady state concentrations for all lysimeters were statistically similar.
3. The leached anions concentrations were statistically similar in all lysimeter types.
4. The leached cations concentrations were statistically similar in all lysimeter types.
5. Cations' concentrations leached at the similar concentrations to wood ash alone.
6. Zinc leached at higher level than either in coal or wood ash, which was caused by the addition of tires to the fuel stream.
7. Lead leached at higher concentrations than coal ash but at similar concentrations to wood ash. Similarity in lead leachability to wood ash arises from the statically similar lead concentration between WT ash and wood ash.

CHAPTER 6

COMPARISON BETWEEN BATCH AND LYSIMETER LEACHING TESTS

Both batch and lysimeter leaching tests have been reported for generating and characterizing leachate from solid wastes (Francis and Maskarinec 1986; Francis et al. 1984; Hjelmar 1990; Jackson and Bisson 1990; Jackson et al. 1984). However, the comparison of these two types of leaching tests in the context of land application of solid waste has not been thoroughly examined. The purpose of this chapter was to compare inorganic contaminant concentrations in batch and lysimeter leaching tests. The comparison will aid in the use of batch leaching tests results in the assessment of risk to ground water contamination (see chapter 7).

The effects of increasing liquid-to-solid ratio on contaminant leaching from WT ash using batch leaching tests were previously examined (see chapter 4). Since, except for alkalinity, there was no statistical difference between the three lysimeter types examined (see chapter 5), lysimeter data presented in this chapter reflects lysimeter type 1 only. Previous research suggest that the best way to compare batch and lysimeter leaching tests is on liquid-to-solid ratio (Kjedsen and Christensen 1990; Van der Sloot 2002; Van der Sloot et al. 1997). Liquid-to-solid ratio is defined as the ratio of the volume of leaching solution to the mass of ash (L/kg) as presented in Equation 6-1.

$$L/S = \frac{V_L}{M_{ash}} \quad (6-1)$$

Where L/S is liquid-to-solid ratio, V_L is the cumulative leachate drained as calculated in equation 5-1, and M_{ash} is the mass of ash in the lysimeter (presented in Appendix ??).

While liquid-to-solid ratios of the lysimeter study varied from 0.1 to 8 L/kg, batch leaching tests were conducted at liquid-to-solid ratios ranging from 3 to 500 L/kg. The comparison between batch and lysimeter leaching tests' conventional parameters and anions concentrations was conducted only at selected liquid-to-solid ratios. Cations and metal concentrations, on the other hand, were graphed as a function of liquid-to-solid ratio.

6.1 Conventional Parameters

Conventional parameters of lysimeter leachate as well as batch leaching tests at selected liquid-to-solid ratios are presented in Table 6-1. It is noted that batch-leaching tests were not carried out at a liquid-to-solid ratio of 1.0 and the highest liquid-to-solid ratio achieved in the lysimeter test is 8 to 1. As presented in Table 6-1, the batch leaching tests pH was similar to lysimeter leachate pH. The pH of both tests (lysimeters and batch) was above 12 for all liquid-to-solid ratio presented. Conductivity was also similar for both batch leaching tests and lysimeters. For example, at a liquid-to-solid ratio of 8 batch leaching tests had a conductivity of 8.0 ms/cm while lysimeters leachate conductivity was 7.6 ms/cm. These results were similar to those presented in the literature. In an experiment conducted on coal ash, at a liquid-to-solid ratio of 1.1, both batch and lysimeter leaching tests leachates had similar pH (9.7) and conductivity (598 mg/cm) (Hjelmar 1990). Similar results were observed by Jackson et al. (1984).

Generally, the lysimeter leachate ORP was negative indicating reducing environments. However, ORP of the batch leaching tests was consistently positive indicating an oxidizing environment. Jackson et al. (1984) contributed the difference in extraction ORP to the vigorous mixing of batch leaching tests in the presence of oxygen

(in head space) keeps the batch leaching tests under oxidized conditions. The effects of head space on the ORP of batch leaching tests was also examined by (Meng et al. 2001).

Table 6-1: Leachate pH, Conductivity and ORP at Selected Liquid-To-Solid Ratios (L/S) for Lysimeter and Batch Leaching Tests

Parameter	L/S (L/kg)	Test Type	
		Lysimeter	Batch
pH	1	12.3	NA
	3	12.4	12.5
	5	12.3	12.1
	8	12.4	12.1
	20	NA	12.1
Conductivity (ms/cm)	1	10	NA
	3	9.5	12
	5	8.0	10
	8	7.6	8
	20	NA	7
ORP (mv)	1	1.7	NA
	3	-32	25
	5	-9.2	20
	8	-100	20
	20	NA	5

Meng et al. (2001) concluded that by decreasing the ratio of head space to leachate volume in batch leaching test, the ORP also decreased. This is especially important when examining metal leachability as will be discussed later.

6.2 Anions

Anion concentrations (mg/L) in batch leaching tests were consistently higher than lysimeter tests as presented in Table 6-2. While concentrations in lysimeter test leachates is thought to represent pore water concentrations, because of the relatively higher liquid-to-solid ratio (often 20:1), concentrations in batch leaching tests are thought to be diluted concentrations (Saranko et al. 1999). At a given liquid-to-solid ratio, anions leached at greater concentrations than in lysimeter tests. For example, while at a liquid-to-solid ratio 5:1 the lysimeter leachate contained on average 13.5, 1,500, and 3.7 mg/L chloride,

sulfate and bromide respectively, batch tests leached 380, 2,800 and 90 mg/L of these cations respectively. In batch leaching tests, these anions are retained in the extraction solution. Thus batch-leaching tests indicate the total solubility of a contaminant and do not provide a rate of mobilization of contaminants. In lysimeter tests, leachate is regularly drained; thus anions from the wash-off phase are removed with every draining which gives an indication of the rate, rather than the total, at which contaminants are leaching (Mudd and Kodikara 2000). This is an important issue, especially when assessing risk associated with the land application of the ash. This point will be further examined in chapter 7.

Table 6-2: Leachate Anions Concentrations Selected Liquid-to-Solid Ratios (L/S) for Lysimeter and Batch Leaching Tests

Parameter	L/S (L/kg)	Test Type	
		Lysimeter	Batch
Chloride (mg/L)	1	207	NA
	3	13.7	400
	5	13.5	380
	8	5.2	264
	20	NA	150
Sulfate (mg/L)	1	1,400	NA
	3	1,500	3,100
	5	1,500	2,800
	8	100	2,500
	20	NA	2,000
Bromide (mg/L)	1	88	NA
	3	4.7	120
	5	3.7	90
	8	4.2	80
	20	NA	60

6.3 Cations

Cation concentrations in batch leaching tests were also higher than lysimeter concentrations as presented in Figures 6-1 and 6-2. At a liquid-to-solid ratio of 3, for example, the batch leaching tests extracted 200-mg/L while the lysimeter leachate

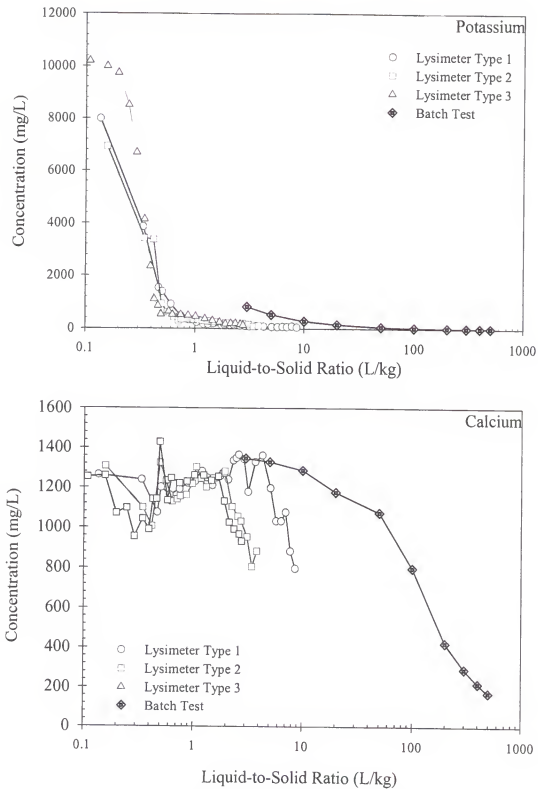


Figure 6-1: Comparison of Potassium and Calcium Concentrations in Batch and Lysimeter Leaching Tests

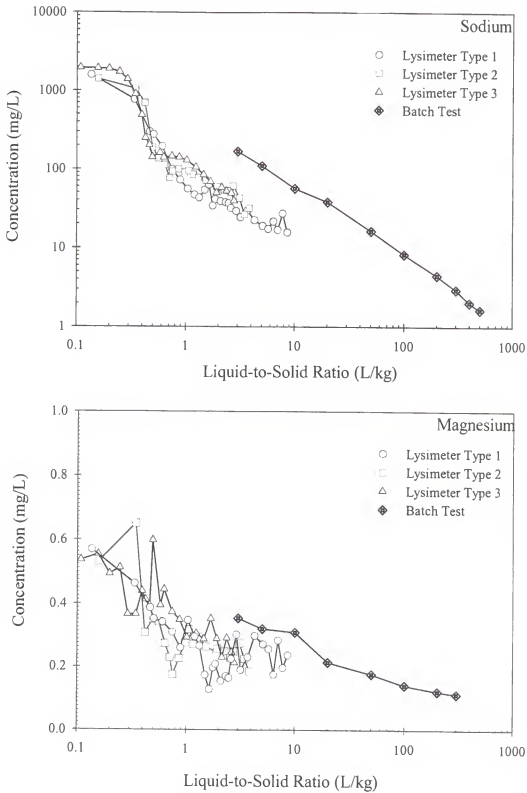


Figure 6-2: Comparison of Sodium and Magnesium Concentrations in Batch and Lysimeter Leaching Tests

contained only 50 mg/L of potassium. This observation was inconsistent with previous research (Hjelmar 1990; Jackson et al. 1984; Mudd and Kodikara 2000). This difference was caused by solid waste examined. The WT ash, examined here, contained large amounts of cations (see chapter 4), as a result the initial wash off phase contained large concentrations of cations. Thus the retention of these anions in the batch-leaching test would cause a large increase over concentrations in lysimeter leaching tests at the same liquid-to-solid ratio. On the other hand, coal ash; the solid waste often examined in the literature, contained lower cations concentrations relative to WT ash. As a result the retention of these lower concentrations in the batch-leaching test did not show any noticeable concentration difference. The calcium concentration in the lysimeter leachate seems to start decreasing around liquid-to-solid ratio of 5 (L/kg). That decrease does not take place in batch leaching tests until liquid-to-solid ratio is around 20 (L/kg) as presented in Figure 6-1. Although batch-leaching tests predicted more cations to leach from the ash at a given liquid-to-solid ratio, initially lysimeter leachate contained higher concentrations than batch leaching tests. For example potassium concentrations in lysimeter type 3 initially were 10,000 mg/L (at liquid-to-solid ratio of 0.1) while the highest potassium concentration in a batch test was only 100 mg/L.

Although, at similar liquid-to-solid ratio, the cations concentrations in the batch leaching test was higher than lysimeter test the cumulative concentration extracted (mg/kg) is the same as presented in Figures 6-3 to 6-6. This observation supports the earlier hypothesis that the retention of the wash off phase, in the batch leaching test, causes the concentrations in these tests to be higher than lysimeter leaching tests. In

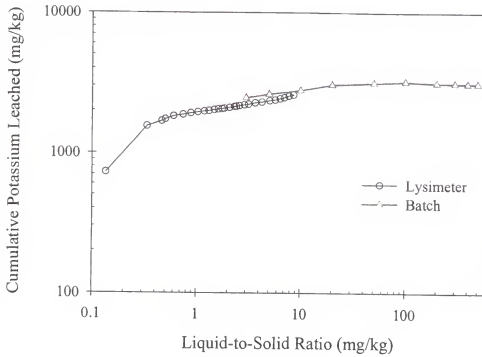


Figure 6-3: Cumulative Potassium Concentration Leached from WT Ash (mg/kg)

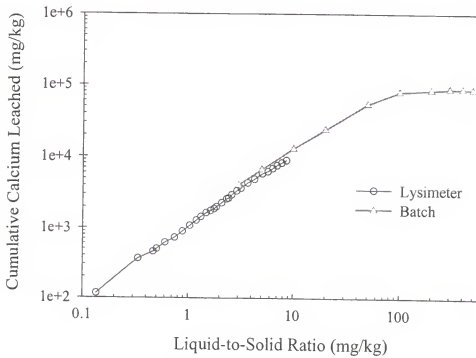


Figure 6-4: Cumulative Calcium Concentration Leached from WT Ash (mg/kg)

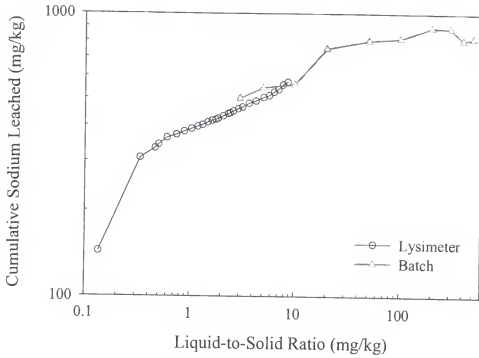


Figure 6-5: Cumulative Sodium Concentration Leached from WT Ash (mg/kg)

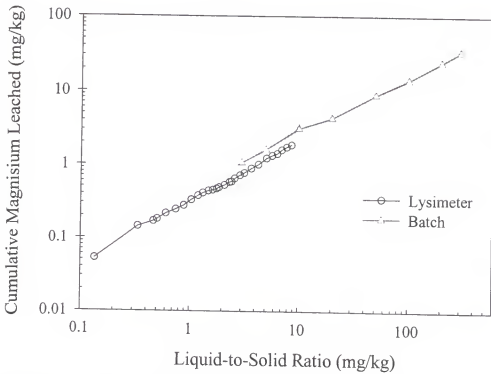


Figure 6-6: Cumulative Magnesium Concentration Leached from WT Ash (mg/kg)

general the cumulative leached anions increase until reaching a liquid-to-solid ratio after which the concentration levels off. At that level the maximum leachable concentration of cations can be estimated. For example, the maximum amount of calcium that can leach per 1 kg of ash is 100g.

6.4 Metals

Batch and lysimeter leachates were analyzed for seven metals (Aluminum, arsenic, barium, chromium, iron, lead, nickel, selenium and zinc). Arsenic, nickel and selenium concentrations were consistently below their detection limit of 5 µg/L in both batch and lysimeter leachates. Although chromium was detected in batch leaching tests (chapter 4), the chromium concentration in the lysimeter leachate was below the detection limit of 1 µg/L. Jackson et al (1984) reported similar findings. This inconsistency was contributed to differences in ORP between batch and lysimeter tests.

In alkaline environment, trivalent chromium (the most common form of chromium) is easily oxidized to hexavalent chromium (Bartlett and James 1988). Hexavalent chromium is not only more toxic than trivalent chromium, it is also more mobile (Bartlett and James 1988). Most of the chromium present initially in the WT ash was most likely in the trivalent form. In the case of the batch-leaching tests, the extraction pH was alkaline and the ORP was consistently above 50 (mv) indicating an oxidizing environment. As a result some of the chromium (predominantly in the trivalent form) might have been oxidized to the hexavalent form and leached as a result. Although the lysimeter leachate pH was also alkaline, the leachate ORP indicated reduced conditions especially during the later part of the experiment. Under these reduced conditions,

chromium would most likely remain in the trivalent form of chromium (III), which is less mobile than Cr (VI) as a result it did leached.

The wash-off phase concentrations were somewhat higher than batch leaching tests concentrations. The iron concentration (Figure 6-7) was as high as 0.35 mg/L but as the liquid-to-solid ratio increased the iron concentration decreased to reach a steady state value around 0.08 mg/L. Similar concentration was observed in batch leaching tests up to a liquid-to-solid ratio of 100 l/kg. Above 100 l/kg, the iron concentration started decreasing steadily as presented in Figure 6-7. It is apparent from the figure that the batch leaching tests simulated the lysimeter tests closely (Hjelmar 1990; Jackson et al. 1984). This is indicated by the similar iron concentrations between the lysimeter and batch leaching tests at a given liquid-to-solid ratio.

Unlike the iron, the barium concentrations continuously decrease as the liquid-to-solid ratio increased (Figure 6-7). Batch leaching test concentrations were similar to lysimeter leaching tests. The barium concentration in the batch leaching tests also continued to decrease as liquid-to-solid ratio increased. The concentration of barium decreased from 0.2 mg/L at a liquid-to-solid ratio of 3 to approximately 0.01 mg/L at liquid-to-solid ratios above 100.

The lead concentration was the lowest at low liquid-to-solid ratios then it increased to reach a maximum concentration of 0.06 mg/L between liquid-to-solid ratio of 3 and 10. After which, the lead concentrations started to decrease again as presented in Figure 5-13. The lead concentration in batch leaching tests was below the instrument's detection limit at liquid-to-solid ratios above 200. The concentration of lead in the batch leaching tests was similar to those present in the lysimeter leachate at similar liquid-to-solid ratios.

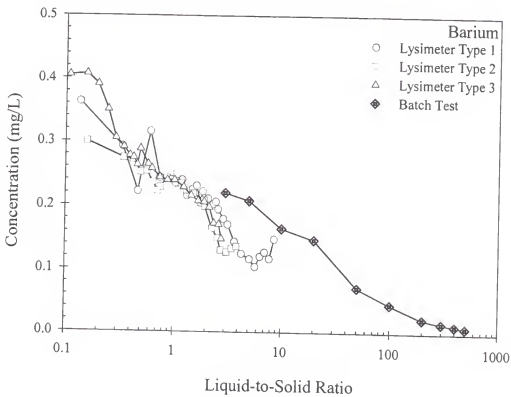
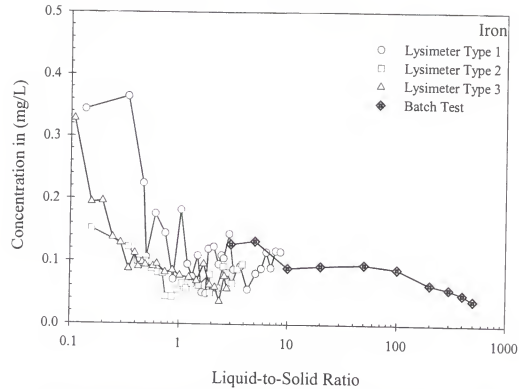


Figure 6-7: Comparison of Metals Concentrations in Batch and Lysimeter Leaching Tests

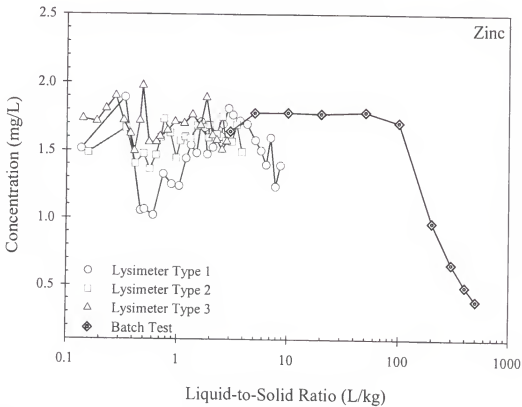
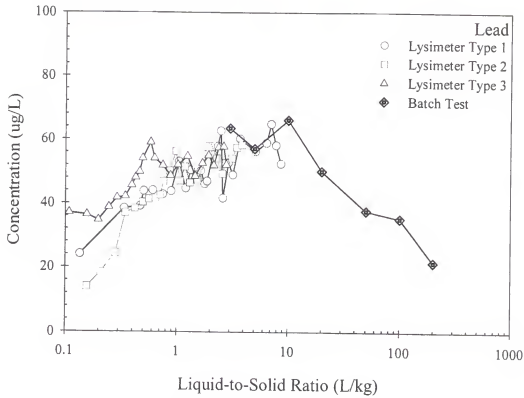


Figure 6-8: Comparison of Metals Concentrations in Batch and Lysimeter Leaching Tests

The zinc concentration remained relatively constant around 1.7 mg/L. Batch and lysimeter leaching tests extracted similar zinc concentrations as presented in Figure 6-8. The zinc concentration started to sharply decrease as liquid-to-solid ratio increased above 100 L/kg. Batch leaching test concentrations for barium, iron, zinc and lead were similar in batch and lysimeter leaching tests. Because of the slow rate of leachability of metals from WT ash, batch and lysimeter leaching tests extracted similar concentrations. Batch leaching tests results were reproducible in lysimeter leaching tests. Further more, these findings validated the normalizing of leaching tests using liquid-to-solid ratio.

When the cumulative metal leached is graphed as a function of liquid-to-solid ratio the lysimeter and batch leaching tests showed high level of correlation. It appears that only barium (Figure 6-9) and zinc (Figure 6-10) reached steady state concentrations

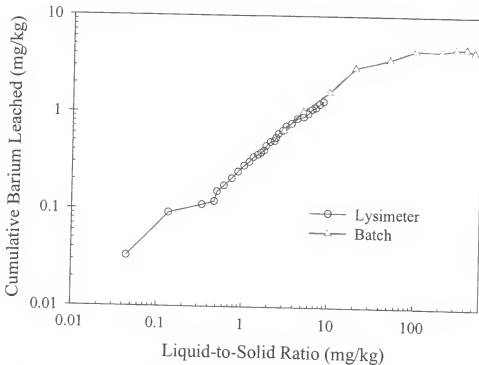


Figure 6-9: Cumulative Barium Concentration Leached from WT Ash (mg/kg)

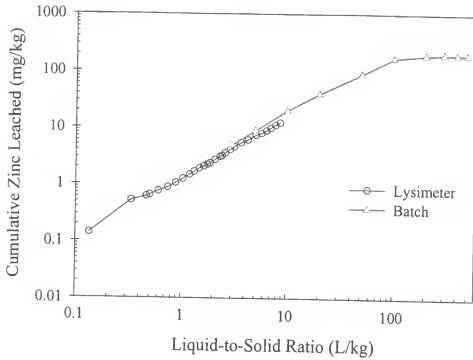


Figure 6-10: Cumulative Zinc Concentration Leached from WT Ash (mg/kg)

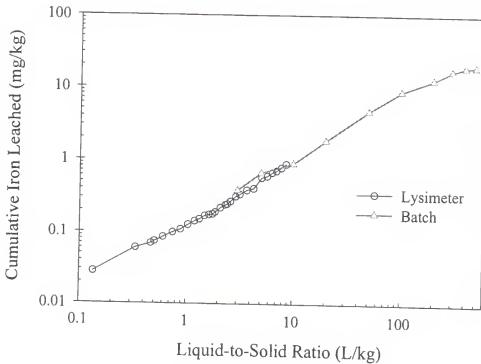


Figure 6-11: Cumulative Iron Concentration Leached from WT Ash (mg/kg)

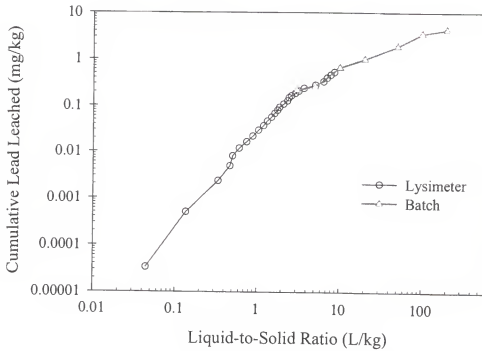


Figure 6-12: Cumulative Lead Concentration Leached from WT Ash (mg/kg)

where the maximum leachable metal content is achieved. The iron (Figure 6-11) and lead (Figure 6-12) concentration did not reach equilibrium concentrations indicating that continued the continued leaching of these metals at higher liquid-to-solid ratios.

Although, atmospheric carbon dioxide did not have a great effect on the pH of the lysimeter leachate compared to batch tests, care must be taken not to neglect its effects on other solid wastes. Carbon dioxide had a minimal effect because of the high buffering available in the WT ash. Wastes with low buffering capacity would most likely be affected. The oxidative state present in batch leaching test compared to the lysimeter, however, may have affected the form of chromium leachability (as discussed earlier). Jackson et al. (Jackson et al. 1984) found that chromium leachability in batch leaching test was greater than lysimeter leaching tests because of the different oxidative state present in the tests.

In general, batch leaching tests over estimated the extractable ion concentrations compared lysimeter tests. This may have been caused by batch leaching test retention of the large ion concentration associated with the wash off phase compared with the lysimeter continual draining. When compared on the basis of total ion extracted batch and lysimeter tests compared well as indicated by the correlation in the graphs. For metals, batch-leaching tests statistically extracted similar concentrations as batch leaching tests. This may have been caused by the small concentrations, compared to ions, which if retained in the batch test would not have effected the concentrations by significant level.

6.5 Summary of the Comparison of Batch and Lysimeter Leaching Tests

- 1 Batch and lysimeter leaching test leachate had similar pH and conductivity.
- 2 Oxidation-reduction potential was consistently positive in batch leaching tests (indicating oxidized environments) and negative in lysimeter tests (indicating reduced environments).
- 3 At a given liquid-to-solid ration batch leaching tests concentrations of both anions and cations were greater than lysimeter concentrations.
- 4 For rapidly leached constituents like anions and cations, the lysimeter leachate was regularly drained, thus ions from the wash-off phase were removed. In batch leaching tests, the ions were retained in the extraction solution causing higher concentrations in batch leaching tests.
- 5 For ions batch leaching test leachate concentrations were higher than pore water concentrations simulated by the lysimeters.
- 6 While chromium was detected in batch leaching tests, it was not detected in lysimeter leachate. This was caused by the differences in ORP between the batch tests and the lysimeter tests.
- 7 At the same liquid-to-solid ratios, the batch and the lysimeter tests had similar concentrations of iron, barium, lead and zinc.
- 8 For slow leached metals leached from WT ash, batch leaching tests leachate concentrations were indicative of pore water concentrations simulated by the lysimeters.

CHAPTER 7

RISK ASSESSMENT TO GROUNDWATER CONTAMINATION OF LAND-APPLIED ASH

Many states have developed policies for determining when a non-hazardous industrial solid waste, such as combustion ash, can be land-applied in a beneficial manner (see section 2.4). These policies typically rely on some form of exposure risk assessment in order to determine whether a specific beneficial use scenario is environmentally acceptable. One part of the assessment process is to assess exposure through the consumption of contaminated groundwater. That exposure is generally examined either by comparing total contaminant concentrations (mg/kg) to a theoretical leaching-based screening limit or by comparing leachable concentrations (mg/L) from a leaching test to risk-based groundwater limits (see section 2.4).

This chapter will evaluate the risk of groundwater contamination from leaching of land-applied solid waste and in particular WT ash, first by using standardized regulatory practices (total metal and SPLP leaching results) then by using the results of the lysimeter leaching tests conducted in conjunction with available contaminant migration modeling software to model the migration of inorganic contaminants in groundwater. By comparing the results of these two approaches the applicability of using batch leaching tests for risk-based decision making will be assessed and the current regulatory risk evaluation techniques will be criticized.

It is unclear if the SPLP concentrations are indicative of pore water concentrations C_L or diluted concentrations at a point down gradient. Since lysimeter tests give an

indication of the actual pore water concentrations, it is possible to use these concentrations in conducting a site specific hypothetical land application scenario. To do so a computer model (MYGRT) is used to assess contaminant concentration in groundwater at varying distances from the application site. The computer model will be used as an example of how a site-specific risk assessment process is applied.

7.1 Risk-based on Partitioning Coefficient Approach

One method to assess risk through the consumption of contaminated groundwater involves the comparison of the total amount of pollutant (mg/kg) present in the waste to a threshold level concentration (e.g., US EPA soil screening level $SSL_{Leaching}$). This threshold concentration represents the maximum pollutant concentration above which contaminant concentration in groundwater would exceed the risk-based groundwater standards. This method utilizes an assumed partitioning coefficient (k_d) and a dilution attenuation factor (DAF) as presented in equation 2-2.

Total metal digestion was carried out on 30 samples. These samples include the 8 ash loads initially collected (each analyzed in triplicate) and the 6 individual WT ash samples subsequently collected (see chapter 3 for more details). Thirteen metals were detected in the ash samples (aluminum, arsenic, barium, cadmium, cobalt, chromium, copper, vanadium, iron, lead, manganese, nickel and zinc). For small sample size (less than 50 samples), the US EPA suggests using a conservative estimate of the arithmetic mean concentration for each contaminant (EPA 2000). This estimate, the 95 percent upper confidence limit (UCL_{95}) of the mean, is used to avoid underestimating the true mean. If the UCL_{95} for a contaminant is less than its risk-based guidance limit (in this case the US EPA soil screening level $SSL_{Leaching}$), then its impact on groundwater contamination is likely (EPA 2001).

Of the 13 metals detected only 6 are assigned a SSL, by the US EPA, for leaching as presented in Table 7-1. The ash samples UCL₉₅ as well as the mean and standard deviation of metal concentrations detected were calculated and presented in Table 7-1. The UCL₉₅ of arsenic, chromium and zinc exceeded their respective SSL for leaching. The UCL₉₅ for chromium (41.4) was slightly higher than the SSL for leaching (38 mg/kg), while zinc (17.4 g/kg) was approximately 3 times greater than its limit (6 g/kg). It is noted that the chromium SSL for leaching is based on hexavalent chromium; no chromium speciation was performed as part of this study. The arsenic UCL₉₅ was 37.6 mg/kg, which is also

Table 7-1: Comparison of Total Metals UCL₉₅ and the EPA's Soil Screening Levels (SSLs) for Leaching

Metal	Units	Mean Concentration ^a	UCL ₉₅ ^a	SSL
Al	g/kg	3.59 ± 1.0	3.90	NA
Fe		29.5 ± 18	35.6	NA
Zn		16.2 ± 3.5	17.4	12
As	mg/kg	35.8 ± 2.1	37.6	29 (5.8)
Ba		32.8 ± 6.0	35.3	1,600
Cd		2.20 ± 0.57	2.40	8
Cr		37.8 ± 10	41.4	38
Co		97.8 ± 36	110	NA
Cu		127 ± 57	147	NA
Pb		52.1 ± 13	56.6	NA
Mn		260 ± 100	300	NA
Ni		13.2 ± 4.6	14.8	130
V		7.14 ± 2.2	7.88	6,000

^a all samples analyzed are included (8 initial ash loads, mixed load, and the six subsequent samples collected).

above its limit (29 mg/kg). However, as of 2006, the drinking water standard for arsenic will be lowered to 10 µg/L. Based on this standard a new SSL for leaching can be calculated for arsenic to be 5.8 mg/kg. The UCL₉₅ for arsenic is 4 times higher than the new SSL for leaching for arsenic. Based on the total metal concentration, WT ash would

pose a risk to groundwater because of its content of arsenic, chromium, and zinc. Of these metals however, arsenic poses the greatest threat since its UCL_{95} is higher than $SSL_{Leaching}$ limit relative to the other metals that exceeded their respective $SSL_{Leaching}$.

7.2 Risk-based on Batch Leaching Test (SPLP) Metal Concentrations

A second method to assess risk to groundwater contamination is by conducting a batch-leaching test (e.g., SPLP). This approach is thought to be better than the previous one since it actually measures leachability rather than assuming it. Since batch leaching tests are carried out at a high liquid-to-solid ratio (20:1), many states guidelines assume that batch-leaching test leachate contaminant concentrations represent actual diluted groundwater concentrations at a compliance point. Thus, it is a common practice in these states to compare the results of a batch-leaching test, such as the SPLP, directly to groundwater standards.

The SPLP leachate contaminant concentrations of 14 WT ash samples (8 initial and the 6 individual subsequent WT ash samples see chapter 2 for more details) were analyzed for the same 13 metals mentioned in the previous section. The concentrations of arsenic, cadmium, cobalt, copper and vanadium were consistently below their detection limits of 5, 55, 1, 5 and 10 $\mu\text{g/L}$ respectively. Again the UCL_{95} on the mean was used as a conservative estimate of the arithmetic mean. Half of the detection limit was used for statistical analysis of concentrations below the detection limits. The instrument's detection limits as well as the number of samples below the detection limit are presented in Table 7-2.

Table 7-2: Comparison of SPLP Metals UCL₉₅ and the Risk-based Drinking Water Standards (DWS).

Metal	N Samples Detected ^a	DL ^b (µg/L)	EPA's DWS (µg/L)	Mean Concentration ^c (µg/L)	UCL ₉₅ ^c (µg/L)
Al	10	70	200 ^c	1,200	2,500
Ba	14	20	2,000 ^d	220	270
Cr	14	1	100 ^d	17	24
Fe	14	20	300 ^c	120	150
Mn	7	11	50 ^c	60	162
Ni	10	15	100	39	90
Pb	14	5	15 ^d	82	100
Zn	14	50	5,000 ^c	1,800	1,900
Cl	14	1.0	250 ^c	200	240
SO ₄	14	1.0	250 ^c	1,800	2,100
TDS	14	-	500 ^c	6,000	6,500
pH	14	1-14	6.5-8.5 ^c	12.1	12.1

^a Number samples analyzed 14

^b DL = Detection Limit

^c Secondary Drinking Water Standard

^d Primary Drinking Standard

^e Include SPLP for the following samples: mixed load and 6 subsequent samples collected

A degree of variability, demonstrated by the difference between the mean and UCL₉₅ concentrations, exists between the 14 samples analyzed. Such variability, which reflects changes in the feedstock and incineration conditions, is expected (see chapter 4 for more details). Of secondary DWS, the UCL₉₅ of aluminum, manganese, sulfate, total dissolved solids and pH were above their respective benchmark concentration. The UCL₉₅ for the leached aluminum (2,500 µg/L) was 12 times higher than the DWS (200 µg/L) while the UCL₉₅ for manganese was only 3 times higher than its DWS (50 µg/L). Both aluminum and manganese are secondary DWS. While aluminum is listed for its aesthetic effect (causes colored water), manganese is listed mainly for its cosmetic effects such as teeth coloration. The UCL₉₅ for sulfate (2,100 mg/L) is 8 times greater than its

DWS. Sulfate gives the water salty taste and may cause diarrhea (EPA 2002b). While the high UCL_{95} for TDS (6,500 mg/L) may increase water hardness and deposits, the high UCL_{95} of pH (12.1) also causes an increase in water deposits.

Lead was the only primary drinking water standard to exceed its UCL_{95} . Lead is a known toxin that causes delays in physical or mental development in children as well as kidney problems and high blood pressure in adults (EPA 1989). The UCL_{95} of the leached lead concentration (100 $\mu\text{g/L}$) was approximately 7 times greater than its DWS limit (15 $\mu\text{g/L}$). The question still remains on whether the SPLP metals concentrations represent pore water or diluted concentrations and what is the degree of dilution if any. If the SPLP leachate concentrations are assumed to be diluted concentrations representative of a drinking water well down gradient, then WT ash land application will pose a risk to human health since one primary and a few secondary DWS were exceeded. If a DAF of 10 is applied the concentrations of manganese sulfate and lead would fall below their DWS limits, while DAF of 20 would also bring all contaminant concentration below their respective DWS limit.

7.3 Risk-based on Lysimeter Results

Because of the low liquid-to-solid ratios in lysimeter tests, lysimeter leachate contaminant concentrations are representative of pore water concentrations C_L . Lysimeter leachate concentrations reflect concentrations detected at point A in Figure 2-1. Of the 13 metals analyzed, 5 metals (aluminum, barium, iron, lead and zinc) were consistently detected in the lysimeter leachate. Since lysimeter results were discussed earlier in great detail (see chapter 5), this section will only concentrate on contaminants that exceeded the DWS (primary and secondary) in the batch tests such as pH, TDS,

sulfate, aluminum, and lead. It was concluded earlier (chapter 5) that depth of ash lift statistically does not have an effect on leachate quality, thus lysimeter type 1 will only be presented in these sections. Contaminant concentrations were plotted as a function of cumulative liquid-to-solid ratio. The cumulative liquid-to-solid ratio could be defined as the ratio of the cumulative leaching solution drained to the mass of WT ash in each lysimeter.

The pH of the lysimeter leachate (Figure 7-1) was above its secondary DWS range (6.5-8.5). This is of particular interest since leachate migrating from a land-applied ash may change the pH of the groundwater enough that it might not be suitable for drinking. The leachate TDS was also above its secondary DWS throughout the duration of the

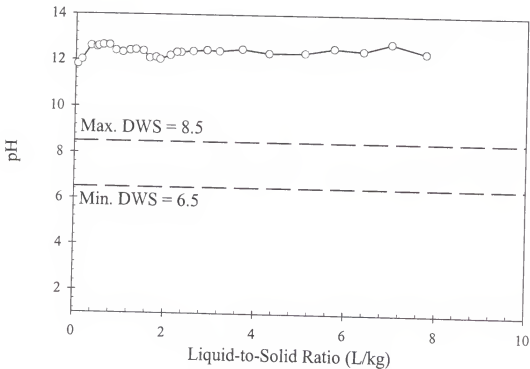


Figure 7-1: Comparison between Lysimeter Leachate pH and DWS

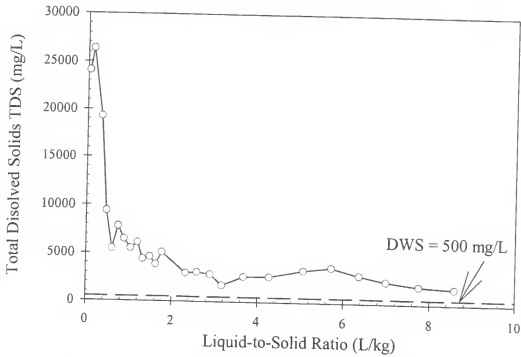


Figure 7-2: Comparison of Lysimeter Leachate TDS Content and DWS

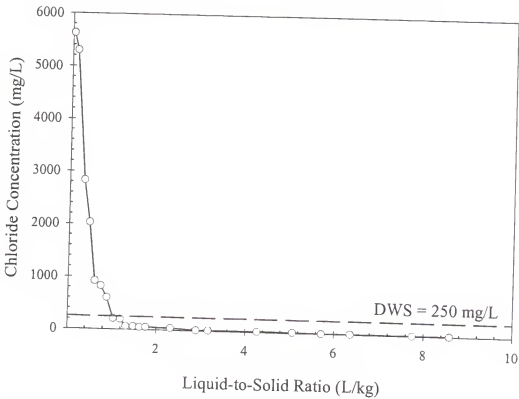


Figure 7-3: Comparison between Lysimeter Leachate Chloride Content and DWS

experiment. Although the TDS in the lysimeter leachate dropped from a maximum of 25,000 to 1,000 mg/L (Figure 7-2), steady state concentrations are still above the secondary DWS. The high TDS in the leachate might lead to undesirable hard water in nearby drinking water wells.

Initially the chloride concentration was high 5,800 mg/L. However, the concentration decreased rapidly to less than the chloride DWS (250 mg/L) as presented in Figure 7-3. The other anion that with concentrations above the DWS was sulfate. The sulfate concentrations in the lysimeter leachate did not decrease as rapidly as the chloride concentration as presented in Figure 7-4. The concentration did not drop below the DWS (250 mg/L) until the end of the lysimeter experiment. The aluminum concentration decreased with increasing liquid-to-solid ratio as presented in Figures 7-5. This is caused

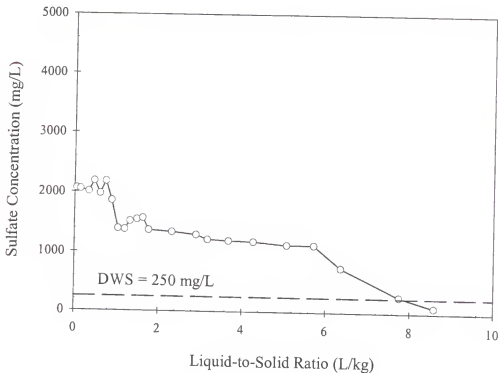


Figure 7-4: Comparison between Sulfate Concentration in Lysimeter Leachate and DWS

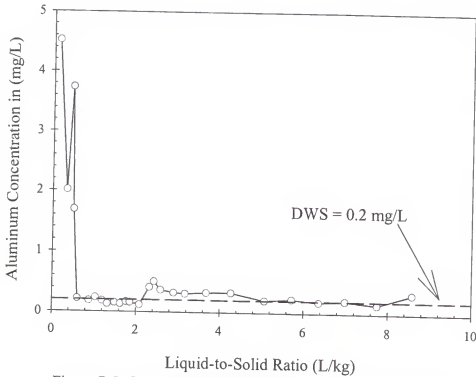


Figure 7-5: Comparison between Aluminum Concentration in Lysimeter Leachate and DWS

by a washing off effect of this metal off mainly ion exchange sites. The initial aluminum concentration (4.5 mg/L) was the maximum concentration leached as presented in Figure 7-5. The aluminum concentration decreased rapidly to hover around the DWS (200 $\mu\text{g/L}$) at liquid-to-solid ratios above 0.4 L/kg.

The only contaminant with a primary DWS that was detected in the lysimeter leachate was lead. Unlike aluminum, the concentration of lead were the lowest at small liquid-to-solid (see chapter 4). The initial leached lead concentration (approximately 6 $\mu\text{g/L}$) was the lowest detected throughout the duration of the experiment. The lead concentrations increased steadily to reach an equilibrium concentration between 50 and 60 $\mu\text{g/L}$ at liquid-to-solid ratios above 2 L/kg as presented in Figure 7-6. Lysimeter results indicate that lead, a primary DWS, is the only metal that leached above its risk-

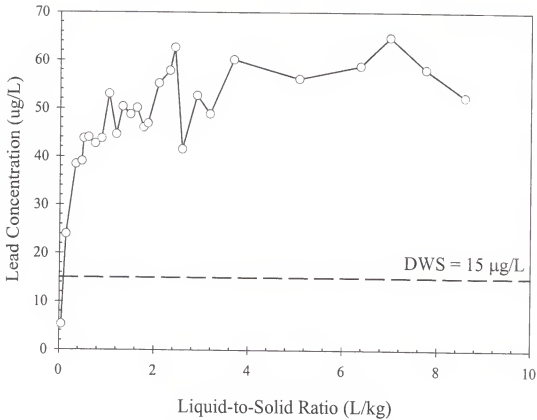


Figure 7-6: Comparison of Lead Concentration in Lysimeter Leachate and DWS based limit. However, these tests also offered an indication of the actual C_L leaching from WT ash and thus facilitated further analysis.

Since lysimeter leachate concentrations give an indication of actual pore water concentrations, DAF may be applied to these concentrations to assess metal concentrations at a compliance point. For example, A DAF of 10 reduces the lead concentrations to levels below its risk-based standard (15 $\mu\text{g/L}$) as well as all the steady state secondary DWS exceeded. That being said, some states (like Florida) may not allow for a mixing zone in land application scenarios. Thus the point of compliance in that case is directly below the ash lift. If that is the case, then the ash can not be land-applied since it leaches lead above the DWS.

7.4 Groundwater Modeling Results for Proposed Scenario

Contaminant concentrations in lysimeter leaching tests were used as input values to model groundwater contamination from the land application of WT ash. The hypothetical scenario examines the migration of contaminant from a land application area of 200m long and wide over a period of 200 years. The computer-modeling program MYGRT was used for the simulation.

The scenario examines the leachate migration of lead, aluminum, zinc, chloride and sulfate from a land-applied area, downward movement through the vadose zone, then into a finite aquifer as explained earlier in section 3.10. MYGRT requires the contaminant concentration to be input as a function of time. To be able to do so, the total volume drained (V_L) from each lysimeter was converted to an approximate time scale by the following equation:

$$t = \frac{V_L}{A * I} \quad (7-1)$$

where V_L is the cumulative volume of leachate collected, A is the area of the lysimeter (0.018 m^2), and I is the assumed infiltration rate (0.3 m/yr). Figure 7-7 presents the concentration of chloride as a function of time as an example. To account for the wash-off phase, the leaching curve was divided into three different zones. The average concentration of each zone was used as input value for the duration of that zone. For example, the chloride concentration of 550 mg/L (average of zone 2) was used as an input chloride concentration for time between 0.15 and 0.35 years. The rest of the values used for chlorides are presented in Table 7-3. In the same manner the concentration of sulfate was also varied as presented in Table 7-3. The lead and zinc concentrations were constant (0.055 and 1.7 mg/L respectively) throughout the leaching duration.

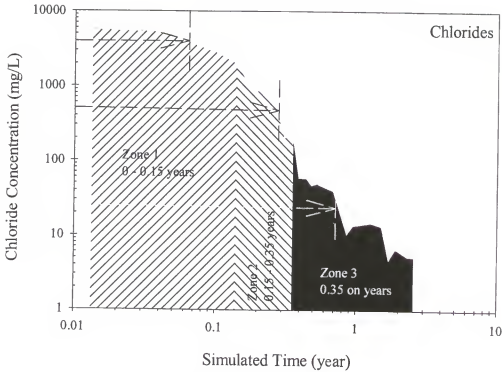


Figure 7-7: Chloride Concentration as a Function of Time in Lysimeter Tests

Table 7-3: Contaminant Concentrations Used by MYGRT

Contaminant	Time Step (year)	Concentration (mg/L)
Cl	0-0.15	4,000
	0.15-0.35	550
	0.35-200	25
SO ₄	0-0.15	2,000
	0.15-0.35	1,500
	0.35-200	250
Pb	0-200	0.055
Zn	0-200	1.7
Al	0-200	0.2

The US EPA used computer models (e.g., EPACMTP) extensively in the development of the risk-based soil screening levels (SSLs) (EPA 1996a; EPA 2001; EPA 2002c). Computer simulations were also used to evaluate appropriate dilution attenuation factors (DAF) used in the risk assessment process. This section presents a simple model that is used as an example of how computer modeling could be applied to determine a site

specific groundwater concentrations or DAF. Input hydrological parameters representative of central (Orlando) Florida were used for this model simulation and were presented in section 3.10 (ref). Model results will vary if input parameters (e.g., hydraulic conductivity) were changed.

7.4.1 Lead

The transport of lead in the subsurface is influenced by complex geochemical interactions (Deutsch 1997). For most risk assessment applications, the US EPA has implemented a modeling approach, which utilizes the MINTEQA2 metals speciation model. Effective sorption isotherms reflecting variations in four geochemical master variables affecting metals fate and transport were used. For lead, the US EPA models predict a k_d value to exceed 2000 L/kg (EPA 1996a). This behavior is typical for lead, which is a strongly sorbed metal, except under low pH conditions (EPA 1996a). The high k_d value results in a long subsurface travel times and pronounced attenuation of concentration values, because much of the lead in soil and groundwater adsorbed on the solid phase. Thus, MYGRT modeling results indicated that the lead did not reach the groundwater within the simulation time (3000 years).

As a conservative assumption and to examine dilution effect only, MYGRT was executed again to calculate the lead concentration with no retardation. The lead concentrations ($\mu\text{g/L}$) of that scenario were plotted as a function of distance from the application site (m) for 5 different time periods (5, 10, 50, 80, and 100 years) as presented in Figure 7-8. Based on an assumed value $k_d = 0$ (with no retardation), it is apparent from the results that lead may pose a risk to human health and the environment since the concentration of lead in the groundwater would increase from 0 (before land application)

to around $22 \mu\text{g/L}$, which is greater than the drinking water standard of $15 \mu\text{g/L}$ (indicated by dotted line on Figure 7-7). This simulation, however, is overly conservative since it did not take into account the retardation and attenuation of the lead on the solid phase.

The lead steady state concentration ($22 \mu\text{g/L}$) is two times less than the pore volume concentration ($55 \mu\text{g/L}$). Based on the assumed input values, MYGRT predicts that a dilution factor of 2 occurs between the application point and a well that is 300 m (1000 ft) away. Based on this hypothetical scenario for lead, applying a DAF of 20 to the pore volume concentrations is not accurate since a DAF of 2 is the maximum

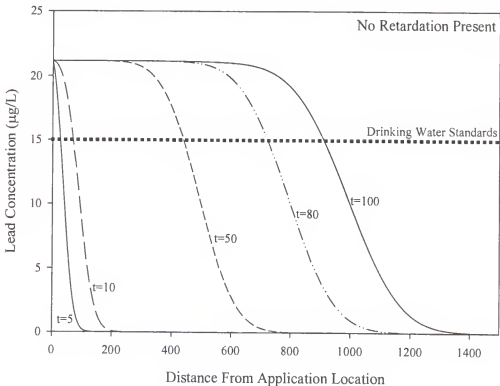


Figure 7-8: Lead Groundwater Concentration Modeled by MYGRT

expected. However, assuming that the SPLP concentration is the diluted concentration at a compliance well is also inaccurate. Again, it is worth noting that the model results will

vary depending on the input values and the purpose of this model was to demonstrate how computer modeling could be applied in the risk assessment process.

7.4.2 Aluminum

As presented in Figure 7-9, the aluminum concentration in groundwater does not exceed the DWS. Thus, based on this simplistic model, the leached aluminum concentration would not pose a use restriction on WT ash.

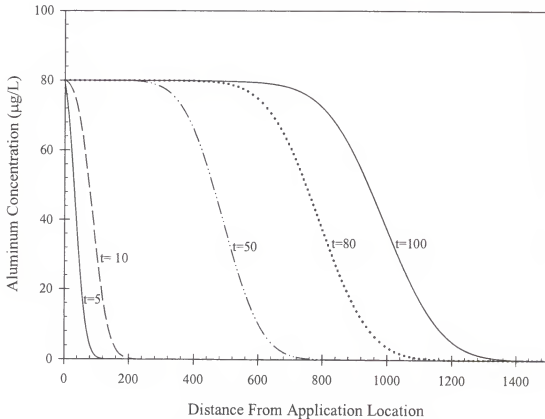


Figure 7-9: Aluminum Groundwater Concentration Modeled by MYGRT

7.4.3 Chloride

Although the initial wash-off phase concentration (4,000 mg/L) was much higher than chloride's risk-based standard, the chloride concentrations modeled in the groundwater was lower than that standard at all times as presented in Figure 7-10. The

maximum chloride concentration predicted (200 mg/L) occurred shortly after land application (5 years) within 100 m from the application site. By examining SPLP results, chlorides concentrations were predicted to pose a use obstacle since their concentrations was above the drinking water standard (250 mg/L). However, as explained earlier (section 5.3), most of the chlorides in the batch leaching tests are cause by the initial flush of chlorides.

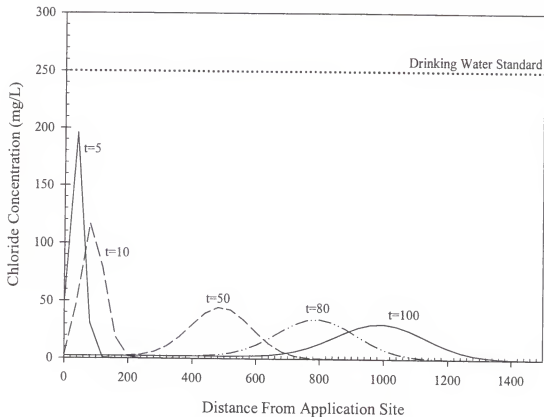


Figure 7-10: Chloride Groundwater Concentration Modeled by MYGRT

It is interesting to note that MYGRT model demonstrated that the wash off phase, in this particular case, did not have an effect on groundwater concentration. This may be the result of the combination of the high hydraulic conductivity and short duration of the wash-off phase. These results, however, may not be correct under all scenarios. An

aquifer with lower hydraulic conductivity may result in a scenario where the chloride concentration in the groundwater would cause a beneficial use limitation.

7.4.4 Sulfate

The sulfate wash-off concentration was high enough so that it affected the concentration in groundwater throughout the duration of the model (3000 years). The

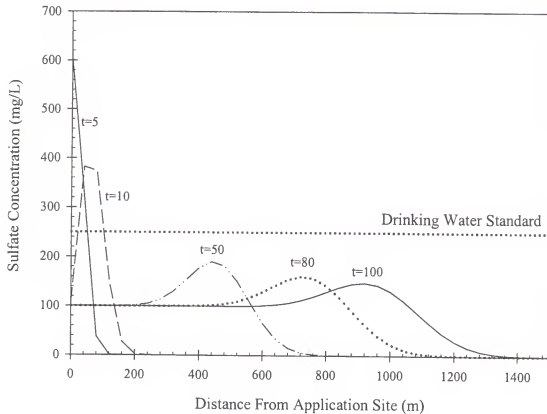


Figure 7-11: Sulfate Groundwater Concentration Modeled by MYGRT

sulfate concentration was above the drinking water standards for 10 years after the application of WT ash as presented in Figure 7-11. The maximum sulfate concentration was diluted 3.3 times, which is less than the 20 to 1 dilution factor suggested by the US EPA (EPA 2001). Unlike chloride, sulfate will pose a problem with groundwater contamination and thus appropriate measures must be taken. Again, it is worth noting

that the model results will vary depending on the input values, and the purpose of this model was to demonstrate how computer modeling could be applied in the risk assessment process.

7.5 Implications and Limitations

- 1 Results presented in this research showed discrepancies between k_d value and batch leaching test approaches used in assessing risk to groundwater contamination.
- 2 Based on the back-calculated metal concentrations, it was concluded that arsenic, chromium, and zinc should leach from WT ash at concentrations that may contaminate groundwater.
- 3 SPLP batch leaching tests results indicate that Arsenic did not leach at detectable level from WT ash. Chromium, and zinc did leach but at concentrations below the DWS.
- 4 The use of the k_d values at a pH of 6.8 overestimates the amount of zinc that might leach. In this scenario the k_d value approach over estimates the amount of metals potentially leaching from WT ash because the k_d used corresponds to a pH of 6.8.
- 5 Based on the SPLP results the UCL_{95} of aluminum, manganese, lead, TDS, and sulfate leached at concentrations above the DWS.
- 6 Based on lysimeter results, aluminum, lead, TDS, chloride and sulfate were above the DWS.

CHAPTER 8
A TIERED APPROACH FOR RISK ASSESSMENT OF GROUNDWATER
CONTAMINATION FROM A LAND-APPLIED SOLID WASTE

**8.1 Outline of Current Methodologies for Risk Assessment Through
Groundwater Contamination**

Some states (e.g., Florida and Massachusetts) have developed methodologies to assess risk through groundwater contamination from the land application of solid waste. These methodologies rely on the conservative assumption that groundwater may be used for direct human consumption. Thus these risk-based standards set groundwater limits above which contaminants concentrations may pose an unacceptable risk to human health if the groundwater is consumed. Overall three different approaches are frequently used by state agencies. These approaches are briefly reviewed below (see section 2.4 for detailed discussion).

One method requires the measurement of the total concentration of a pollutant (mg/kg) present in the waste and compares it to a back calculated theoretical value above which a pollutant that may leach into solution. This approach allows for affordable testing since only total metal analyses is needed, thus the results can be directly compared to risk-based standard for leaching (i.e., $SSL_{Leaching}$) presented in the SSG and various state guidelines (see section 2.4 of this document). This approach, however, is largely dependent on the estimation of k_d values.

A second method to assess risk to groundwater contamination is by conducting batch-leaching tests (e.g., SPLP). This approach is thought to be better than the previous one since it measures waste leachability rather than assuming it. Since batch leaching

tests are carried out at a high liquid-to-solid ratio (20:1), some state guidelines assume that the batch-leaching test concentrations represent diluted groundwater concentrations at a compliance point. Thus, it is a common practice in these states to compare the results of a batch leaching test, such as the SPLP, directly to groundwater standards. Contrary to this, the EPA assumes that the SPLP leachate contaminant concentrations measured are those encountered in pore volume as the leachate leaves the land-applied solid waste (EPA 2001). As a result, the EPA recommends applying a DAF to contaminant concentrations in the SPLP leachate. Depending on whether a DAF is used in the risk assessment process, a land-applied solid waste may or may not pose a risk to groundwater contamination.

The third alternative is the use of lysimeter leaching tests (see section 2.4). These tests permit the simulation of waste leachate concentrations closer to that encountered in actual land-applied waste conditions. Although no accepted standardized protocol is available for these test methods in the US, some states accept results from lysimeter studies on a case-by-case basis as they generate more concentrated leachate that is indicative of pore water concentration. By applying the appropriate DAF to the lysimeter leachate concentrations, an estimate of groundwater concentrations can be achieved. The main short coming of lysimeter leaching test is the long time required to conduct the test and the associated cost.

8.2 Problems Associated with Current Methodologies of Risk Assessment to Groundwater Contamination

Previous chapters presented in this dissertation highlighted the problems associated with the current methodologies used to assess risk through groundwater contamination from land-applied solid wastes. The use of total metal concentrations and a partitioning

coefficient to assess risk through groundwater contamination from a land-applied solid waste was shown to be inaccurate since it utilizes k_d values established for soils that might not be correct for the particular solid waste of interest. Using this approach, WT ash was predicted to leach arsenic, chromium and zinc at concentrations above the risk-based limits. That was not consistent with results measured using batch leaching tests. Although chromium (0.024 mg/L) and zinc (1.9 mg/L) leached from WT ash when leached using the SPLP, their concentrations were below the risk-based standards of 0.1 and 5 mg/L respectively. Arsenic was not even detected in the leachate. Consequently, the partitioning coefficient approach should not be used for assessing risk to groundwater contamination from land-applied solid wastes like WT ash.

Using batch leaching tests to assess risk to groundwater contamination does not give an indication of the degree that dilution occurs in the leaching test. While applying a DAF may underestimate the contaminant concentration at the compliance point, not applying any DAF may also be overly conservative. In the scope the research presented earlier, what the SPLP leachate concentrations represent was unclear. The conclusion that the SPLP concentrations of some contaminants (i.e., lead and zinc) actually represent pore water concentration was not known until other non-standardized tests (i.e., liquid-to-solid ratio and lysimeter tests) were performed. Thus, there is a need to modify the some of the methods used to assess risk via groundwater contamination to alleviate some of the uncertainties present in the current methodologies.

8.3 Proposed Approach for Risk Assessment

Uncertainty in assessing groundwater contamination from land-applied solid wastes makes it difficult to allow land application of solid wastes over a wide range of use scenarios. The underlying question that the risk assessment process should answer is

“What is the potential for contaminant release and the risk associated with that release from a given solid waste under the selected management scenario?” The risk assessment methodology for answering these questions should be consistent under a wide array of applications and beneficial use scenarios. From the discussion in the previous two sections (8.1 and 8.2), it is obvious that the current risk assessment methodology followed by some states does not answer that question consistently. Consequently, there is a need for a risk assessment methodology that is flexible enough to answer that question accurately.

Kosson et al. (2002) proposed a framework for evaluation of leaching in solid waste management. However, that framework encompasses a wide array of beneficial use scenarios for solid wastes (Kosson et al., 2002). As a result, it evolved many tests that may not be needed in the risk assessment process and an associated increase in cost. The risk assessment approach proposed in this section allows for the most accurate risk assessment process of only one particular scenario (land application of solid waste). This allows for a focused approach that would keep the testing protocols simple and quick.

The goals of the proposed approach are to:

- 1 Provide conservative and realistic estimate of contaminant release from land application of solid wastes.
- 2 Utilize existing testing techniques that can be carried out in a relatively short time.
- 3 Provide flexibility to allow evaluations to be based on the regulator requirements.
- 4 Be cost effective.

To achieve the above objectives, a tiered approach for evaluating contaminant release is proposed. Each successive tier provides information that is more specific to the material being tested than the previous tier. If time or economic considerations do not

justify more detailed evaluation, the results of a single test can also be used as the most conservative approach for management decisions. Moving through the different tiers provides an increasingly more realistic and less conservative estimate of contaminant releases, but also requires more extensive testing.

Three tiers or levels of assessment can be defined to address most solid waste land application scenarios. Tier 1 is a screening based assessment and is the methodology that is currently followed by some states. Tier 2 attempts to measure C_L by conducting a relatively quick and cheap batch leaching test. Tier 3 measures C_L by conducting a more extensive leaching test that assesses release under environmental conditions. Figure 8-1 shows a flow diagram of the proposed approach.

8.3.1. Tier 1

This level involves a screening test that provides the most conservative estimate of contaminant release. This screening level would require the test to be completed in less than one day. The test is outlined as follows:

Synthetic Precipitation Leaching Procedure (SPLP): This test provides a conservative estimate of the leachable metal concentration. It is conducted at a 20:1 liquid-to-solid ratio. The extraction pH is governed by the pH of the solid waste. Consequently, this regulatory batch-leaching test gave the best estimate of metal leachability under beneficial use scenarios. The concentrations measured by the SPLP should be compared directly to risk-based groundwater limits (i.e., DAF of 1). For example, in Florida concentrations measured by the SPLP are compared directly to the drinking water standards.

If the results of tier 1 analysis indicate that contamination to groundwater may occur when the solid waste in question is land-applied (i.e., SPLP concentrations are

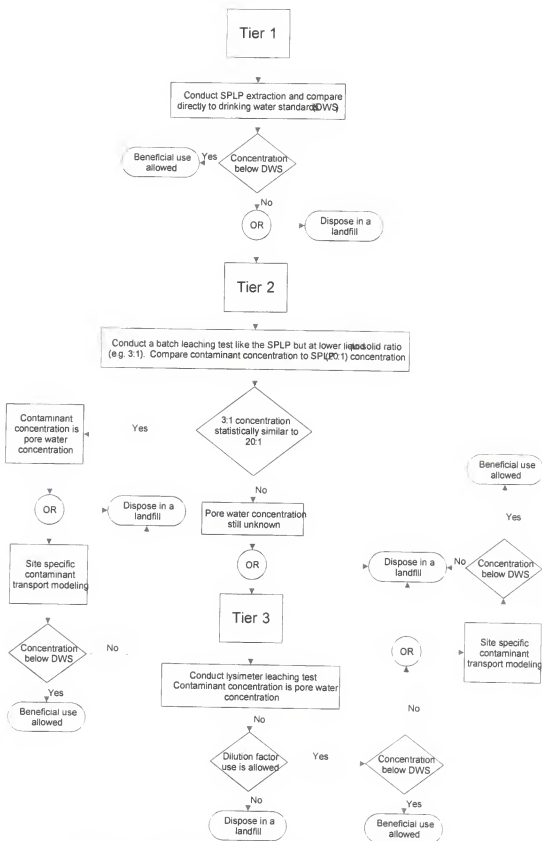


Figure 8-1: Schematic of the Proposed Tiered Approach

greater than risk-based groundwater standards), and if a less conservative and more tailored approach to estimate contaminant release is desired, then tier 2 analyses can be conducted.

8.3.2 Tier 2

This level provides a less conservative estimate of release than the previous tier. This level examines leachability as a function of liquid-to-solid ratio. As discussed briefly in section 8.1 and 8.2, the use of batch leaching tests to assess risk via groundwater contamination does not give an indication of the degree of dilution occurring during the leaching test. In another word, it is unclear if the concentrations measured by the SPLP represent C_L or a diluted concentration down gradient in the groundwater. Thus, applying a DAF may under estimate the contaminant concentration at the compliance point and not applying a DAF may be overly conservative. This raises the need to measure the actual pore water contaminant concentrations. The test proposed in this section can be used to approximate pore water concentrations as well as initial leachate composition. The test involves using the SPLP extraction, however, at a low liquid-to-solid ratio.

As the liquid-to-solid ratio of a batch leaching test decreases, contaminant concentrations increase as the leachate becomes more concentrated. The concentrations continue to increase until reaching a point where a decrease in liquid-to-solid ratio does not cause an increase in concentration as presented in Figure 8-2. At that liquid-to-solid ratio, contaminant concentrations are representative of the actual pore water concentrations C_L . By conducting a batch-leaching test at a low liquid-to-solid ratio (e.g., 3:1 proposed here), there is more likelihood that the contaminant concentrations are pore water concentrations. Lower liquid-to-solid ratios may be more appropriate, but for some

material a liquid-to-solid ratio less than 3:1 is difficult to filter due to liquid-solid separation limitations.

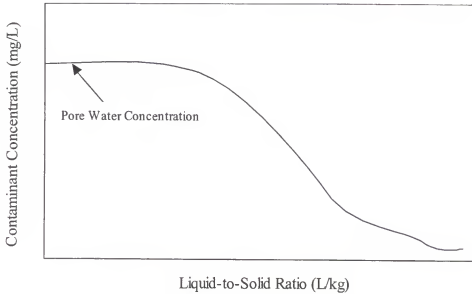


Figure 8-2: Effect of Liquid-to-Solid Ratio on Contaminant Concentrations

Leachability as a Function of Liquid-to-Solid Ratio: The objective of this test is to determine the contaminant concentrations in the pore water volume C_L . By comparing contaminants concentrations in the 3:1 liquid-to-solid ratio batch leaching test to the concentrations of the SPLP extraction a better risk assessment via groundwater contamination from land application of the solid waste can be made as follows:

- 1 If the contaminant concentrations in the 3:1 batch-leaching test are higher than the concentrations in the SPLP extraction test, then it is still uncertain if the contaminant concentrations in the 3:1 batch leaching test is representative of pore water concentration or diluted ones. Risk assessors cannot apply the appropriate dilution attenuation factor to approximate contaminant concentrations at the point of compliance. Thus, progressing to tier 3 may be needed or the solid waste can be appropriately disposed of in a landfill.
- 2 If contaminant concentrations in the 3:1 batch-leaching test are similar to SPLP concentrations, then these concentrations are indicative of pore water concentrations. One of the following actions can be taken:

- a. If a dilution factor can be applied to the SPLP concentrations (based on the location of the point of compliance), then the diluted concentrations are compared to the appropriate risk-based groundwater standards.
 - i. If the diluted concentrations are above the risk-based groundwater standard, then a site specific contaminant transport model could be used in conjunction with the pore water concentrations or the solid waste can be appropriately disposed of in a lined landfill.
 - ii. If the diluted concentrations are below the risk-based groundwater standard, then land application of the solid waste does not pose an unacceptable risk to human health and the environment. Land application is allowed.
- b. If a dilution factor cannot be applied to the SPLP concentrations (meaning the pore water concentrations as the leachate leaves the solid waste has to be below the risk-based groundwater standard), then the land application of the solid waste causes an unacceptable risk to human health via groundwater contamination. Appropriate disposal in landfill scenario is required.

If tier 2 analysis is still inconclusive about the concentrations in the pore water volume as the leachate leaves the land-applied solid waste, then it may be economical to proceed to tier 3. Unlike tier 2, which is relatively quick, tier 3 requires more time.

8.3.3 Tier 3

This tier offers the greatest amount of testing and evaluation. The test method proposed for this tier is the lysimeter leaching test (see chapter 3 for details). This method examines leachability for compacted materials and closely resembles actual field conditions of the land-applied solid waste. Since lysimeter leaching tests are carried out at such a low liquid-to-solid ratio (<1), contaminant concentrations in the leachate resemble concentrations of pore water. Lysimeter leachate contaminant concentrations can be used in conjunction with a dilution attenuation factor to approximate groundwater concentrations at the point of regulatory point of compliance. Computer models (i.e., MYGRT, EPACMTP) can also use lysimeter leachate contaminant concentrations and

site-specific hydrological parameters to model contaminant flow in groundwater.

Computer models better account for actual mixing in groundwater and attenuation with site specific parameters. This process however, is costly. Tier 3 analysis is conducted as follows:

- 1 Contaminants concentration in the lysimeter leachate is measured. These concentrations are indicative of pore water concentrations.
 - a. If a dilution factor can be applied to these concentrations (based on the location of the point of compliance), then the diluted concentrations are compared to the appropriate risk-based groundwater standards.
 - i. If the diluted concentrations are above the risk-based groundwater standard, then a site specific contaminant transport model could be use in conjunction with the pore water concentrations measured or the solid waste can be appropriately disposed of in a lined landfill.
 - ii. If the diluted concentrations are below the risk-based groundwater standard, then land application of the solid waste does not pose an unacceptable risk to human health and the environment. Land application is allowed.
 - b. If a dilution factor cannot be applied to lysimeter concentrations (meaning the pore water concentrations as the leachate leaves the solid waste has to be below the risk-based groundwater standard), then the land application of the solid waste causes an unacceptable risk to human health via groundwater contamination. Appropriate disposal in landfill scenario is required.

8.4 Approach Limitations

The proposed approach assesses risk associated with the consumption of contaminated groundwater, thus direct exposure should be addressed prior to the implementation of this approach. Total contaminant concentrations should not exceed the direct exposure limits outline earlier in section 4.2. The methodology proposed assesses risk associated with a land application scenario. Although the process gives a more accurate estimate of groundwater contamination, it does not take into account changes in pH. For some poorly buffered solid wastes, there may be a change in pH

associated with the constant percolation of rainwater through the waste. At low pH, as presented in section 4.4, metals tend to be readily released from the solid waste. Thus, for poorly buffered solid wastes pH changes may be need to be considered.

The methodology presented also relies on the assumption that equilibrium is reached within the allowed contact time of 18 hours. That may not be true in all cases. Longer contact time may be required in some cases where contaminant release is expected to be slow. Finally lysimeter leaching tests may represent different extraction conditions than batch leaching tests. These different conditions may lead to conflicting results between tier 2 and tier 3 analysis. Thus, care must be taken to address these issues.

CHAPTER 9 CONCLUSIONS

9.1 Summary

This dissertation presented an evaluation of methodologies used to assess risk to human health and the environment from groundwater contamination via leaching of contaminants from land-applied solid waste. This was accomplished through the detailed characterization of one particular solid waste (wood and tire ash). The characterization included total metal analysis and an examination of inorganic contaminant leachability using regulatory batch leaching tests. It is important to remember that this research was not intended to be a beneficial use demonstration study; rather it examines different methods applied in risk-based solid waste management decision-making. This section summarizes the results of this research while the following section presents the final conclusions.

The initial characterization of WT ash indicated the ash was not a toxicity characteristic hazardous waste. The ash did, however, contain some heavy metals (i.e., arsenic and lead) at concentrations high enough to pose a risk to human health and the environment if they leach into groundwater. The lead concentration in the SPLP extraction solution was approximately 53 µg/L. Zinc was also detected at 1.7 mg/L in the SPLP. Unlike lead, which leached above its DWS (15 µg/L), the zinc did not leach above the DWS (5 mg/L). The effects of change in pH, liquid-to-solid ratio, and contact time were also examined. Of these factors, pH was shown to be the most controlling.

Lysimeter leaching tests were also conducted to better examine contaminant release from the ash under environmental conditions. The low liquid-to-solid ratio and the slow downward movement of the leaching solution through the ash in a lysimeter test simulate environmental conditions better than a batch leaching test. The effect of increase in the thickness of the ash lift in a land application scenario was also examined. This was accomplished by using lysimeters containing different thickness of ash (1, 2 and 3 feet ash lifts). In general, the depth of the ash did not have an effect on the leachate quality. Contaminant concentrations in the lysimeter leachate were also compared to concentrations in batch leaching tests. Batch leaching tests statistically extracted larger concentrations of ions than lysimeter leaching tests at similar liquid- to-solid ratio. However, both lysimeters and batch leaching tests extracted statistically similar metal concentrations. Based on total leached mass both lysimeter and batch leaching tests extracted similar contaminant concentrations (mg/kg).

Standardized regulatory practices used to assess risk to groundwater contamination from the land application of the ash were examined. Based on total metal content (mg/kg) comparison with the theoretical risk-based standard for leaching (mg/kg), arsenic, chromium and zinc were found to pose a risk to groundwater contamination. When the SPLP contaminant concentrations were directly compared to the DWS, lead ($UCL_{95} = 100 \mu\text{g/L}$) was the only contaminant to exceed its risk-based standard. When contaminant concentrations in the lysimeter leachate were compared to the DWS, aluminum, pH, chlorides, and sulfate exceeded the secondary DWS. The lead concentration was also above the primary DWS.

A hypothetical land application scenario was proposed. The contaminant concentrations in lysimeter leachate were used in conjunction with site-specific parameters to model groundwater contamination using the computer software MYGRT. A hypothetical land application scenario was proposed in an area near Orlando, Florida. The model predicted that, for the most part, the land application of the ash does not pose an unacceptable risk to groundwater contamination since there is enough attenuation in the soil and dilution in the groundwater to reduce the contaminant concentrations to below the risk-based DWS.

Because of the uncertainties associated with the current risk assessment methodologies, an alternative approach was proposed. The main premise behind the new approach is to approximate the contaminant concentrations in the pore water (C_L) as it leaves the land-applied solid waste. C_L was measured by conducting a modified SPLP extraction test at a lower liquid-to-solid ratio (i.e., 3:1), or by conducting lysimeter leaching test. By approximating C_L risk, either a dilution factor or contaminant transport model may be applied to estimate the contaminant concentration at the point of compliance. The following section presents detailed conclusions and major findings of this research.

9.2 Conclusions

The following section presents detailed conclusions and major findings of this research.

- 1 The concentrations of most inorganic constituents of WT ash were similar to wood ash alone concentrations presented in the literature, but WT ash has higher concentrations of iron and zinc as a result of the tires.
- 2 When compared to direct exposure risk-based limits, iron exceeded Florida's SCTL for residential setting while arsenic exceeded Florida's SCTL as well as US EPA soil screening level.

- 3 TCLP metal concentrations for the WT ash were below the RCRA toxicity limit. Thus the WT ash examined here was not a toxicity characteristic hazardous waste.
- 4 Of the batch leaching tests evaluated the TCLP, SPLP and DI water extracted statistically similar concentrations of inorganic constituents. The similarities between these tests may have been caused by the high ionic strength that could have caused a reduction in the acetic acid chelating ability.
- 5 When compared to the WET, the TCLP extracted lower concentrations of most inorganic constituents, but similar concentrations of barium, calcium and lead. The WET calls for citric acid, which has better chelating ability than the acetic acid used by the TCLP.
- 6 Extraction pH was the most controlling factor governing metal leachability. The concentration of most metals was at its maximum at pH values less than 4 and minimum at pH values between 9 and 10. The concentration of some metals (e.g., lead and zinc) increased again at pH values greater than 11. The increase was most likely caused by the ability of these metals to form soluble metal hydroxides at alkaline pH values.
- 7 The change in liquid-to-solid ratio had the greatest impact on the concentration of potassium and sodium. The concentrations of lead, chromium and zinc were not as affected by the change in liquid-to-solid ratio. The concentrations of lead, chromium and zinc were most likely solubility controlled, thus the change in liquid-to-solid ratio did not change their concentrations.
- 8 Steady state concentrations were achieved within the 18 hours prescribed by the SPLP and the TCLP.
- 9 Steady state concentrations were statistically similar in the three-lysimeter types examined (0.3, 0.6 and 0.9 m WT ash lift). Because of the slow movement of the leachate through the lysimeter, adequate time was allowed for the contaminants to reach similar concentrations regardless of the lift thickness.
- 10 The leached lead and cations concentrations in the lysimeter tests were similar to that leached from wood ash alone values presented in the literature.
- 11 The concentration of zinc in lysimeter leachate was higher than that extracted from wood ash alone.
- 12 When compared to lysimeter leaching tests, at similar liquid-to-solid ratios, batch-leaching tests extracted greater concentrations of anions and cations. In lysimeter tests, leachate is regularly drained; thus ions from the wash-off phase were removed with every draining, while the wash-off phase was retained in batch leaching tests. Based on cumulative leached mass both lysimeter and batch leaching tests extracted similar contaminant concentrations.

- 13 Both batch and lysimeter leaching tests extracted similar concentrations of heavy metals, but, because of the different ORP conditions, chromium was detected in batch and not lysimeter leaching tests.
- 14 Based on the risk assessment using k_d value approach, arsenic, chromium and zinc pose a risk to human health and the environment through groundwater contamination. However, leaching tests did not indicate that these metals leach above their respective groundwater standards. Thus this approach may not apply for solid wastes since most of the theoretical k_d values used were developed for soils at a pH of 6.8.
- 15 Based on risk assessment using the SPLP batch leaching test compared directly to the DWS, aluminum, manganese, sulfate, total dissolved solids and pH exceeded their secondary DWS. Of the primary drinking water standards, lead was the only contaminant to exceed its risk-based DWS.
- 16 Based on the risk assessment using lysimeter leaching tests, aluminum, pH, chlorides and sulfate exceeded their respective secondary DWS. Lead exceeded its primary DWS.
- 17 The hypothetical reuse scenario modeling results, indicated that if there is enough attenuation in the soil and dilution in the groundwater, the land application of the WT ash may not pose an unacceptable risk via groundwater contamination.

9.3 Needs for Future Research

Research presented here and in the literature indicated the possibility of lead contamination of groundwater from wood ash. Often, natural wood does not contain contaminants at a level that may cause a risk, however, during the combustion process some trace elements are concentrated in the ash and may pose a risk if improperly managed. Research is needed to examine the potential leachability of contaminants (i.e., lead) from wood ash alone. Chromium leachability in the batch leaching test may indicate a possible oxidation of the chromium to the hexavalent form. Chromium, speciation needs to be conducted before land application of WT ash is allowed.

The failure of the TCLP and to a lesser extent the WET to chelate metals needs to be further investigated. New toxicity characteristic tests may be needed to address the effect of high ionic strength on the chelating ability of organic acids. The use of the

theoretical k_d values to examine risk via groundwater contamination was found to be inaccurate, thus there may be a need to develop new waste-specific k_d values. For example, k_d values for alkaline ashes, or k_d values for construction and demolition fines.

Although reliance on a single waste management scenario as the basis for leach testing may simplify the process of risk assessment, examination of the leaching potential of wastes as actually managed is needed to better assess the risk posed by beneficial use option of that waste. Chapter 8 presented an approach to evaluate the risk associated with the land-application scenario. The approach has been developed based on the result of contaminant leaching from one solid waste (highly buffered alkaline ash). Thus, there is a need to validate the approach to encompass a wide variety of solid wastes and examines the applicability of the proposed leaching tests to different beneficial uses of solid wastes

The validation would require the solid wastes potentially examined to undergo all three tiers of testing. Large field-scale studies must also be conducted to ensure the accuracy of the risk assessment process. Interactions occurring at a lab-scale level may not be similar to those encountered in the field, thus there is a need to expand the testing to include large-scale field studies. This research targeted beneficial use through the land application of solid wastes, other beneficial use scenarios that might be different (i.e., mine fill, road base) and need to be examined also.

APPENDIX A
COMPOSITE ASH SAMPLE MIXING RATIOS

Table A-1: Contribution of Ash Loads to Composite Ash Sample

Load Number	% by Mass
1	11%
2	13%
3	13%
4	12%
5	13%
6	12%
7	14%
8	13%

APPENDIX B
SAMPLE MASS FOR TOTAL METAL ANALYSIS

Table B-1: Samples Mass Used for Total Metal Digestion

Load #	Sample #	Sample Wet Mass	Average Load Moisture Content	Sample Dry Mass
Load 1	#1	1.95	26%	1.43
	#2	2.01		1.48
	#3	2.09		1.53
Load 2	#1	2.00	28%	1.45
	#2	2.18		1.58
	#3	1.98		1.43
Load 3	#1	2.02	29%	1.44
	#2	1.94		1.38
	#3	1.92		1.37
Load 4	#1	1.92	28%	1.39
	#2	2.58		2.58
	#3	1.95		1.95
Load 5	#1	2.04	28%	1.47
	#2	2.19		2.19
	#3	2.05		2.05
Load 6	#1	2.04	31%	1.41
	#2	2.01		2.01
	#3	2.06		2.06
Load 7	#1	2.03	31%	1.39
	#2	1.94		1.94
	#3	1.94		1.94
Load 8	#1	2.05	31%	1.42
	#2	2.71		2.71
	#3	1.89		1.89
Mixed Load	#1	2.04	29%	1.45
	#2	2.10		2.10
	#3	2.01		2.01
Load 1	Dup	1.96	26%	1.43
Load 1	MS	1.98		1.45
Load 1	MSD	2.02		1.49

APPENDIX C
WOOD AND TIRE ASH MOISTURE CONTENT

Table C-1: Moisture Content of Different Ash Loads

Load #	Mass of Empty Crucible (g)	Mass of Crucible and Wet Ash (g)	Mass of Crucible and Dry Ash (g)	Mass Water (g)	Moisture Content	Average %MC	Stdev.
Load 1	1.0054	20.9343	15.6784	5.2559	0.2637	26.4%	0.2%
	1.0045	20.7974	15.6028	5.1946	0.2624		
	1.0058	22.8633	17.0368	5.8265	0.2666		
Load 2	1.0580	30.0557	21.7052	8.3505	0.2880	27.6%	1.8%
	1.0550	24.9140	18.1210	6.7930	0.2847		
	1.0567	31.6631	23.8330	7.8301	0.2558		
Load 3	1.0108	23.5518	17.0924	6.4594	0.2866	28.6%	2.3%
	1.0094	28.8108	20.2291	8.5817	0.3087		
	1.0089	36.6916	27.2886	9.4030	0.2635		
Load 4	1.0545	25.4691	18.4344	7.0347	0.2881	27.6%	2.1%
	1.0541	28.2850	21.4203	6.8647	0.2521		
	1.0575	27.2916	19.7276	7.5640	0.2883		
Load 5	1.0579	32.0533	24.1159	7.9374	0.2561	27.8%	1.9%
	1.0539	26.6395	19.2327	7.4068	0.2895		
	1.0534	23.6416	17.1094	6.5322	0.2892		
Load 6	1.0080	31.2861	21.8855	9.4006	0.3105	30.8%	0.5%
	1.0108	28.7086	20.0908	8.6178	0.3111		
	1.0403	29.9269	21.1909	8.7360	0.3024		
Load 7	1.0096	30.5749	22.2152	8.3597	0.2828	31.2%	2.6%
	1.0092	31.3235	21.3995	9.9240	0.3274		
	1.0074	27.8870	19.1088	8.7782	0.3266		
Load 8	1.0086	24.7499	17.5180	7.2319	0.3046	30.6%	1.4%
	1.0110	29.1858	20.9265	8.2593	0.2931		
	1.0076	20.8071	14.4530	6.3541	0.3209		
Mixed Load	1.0045	20.7974	15.6028	5.1946	0.2624	29%	1.2%
	1.0567	31.6631	23.8330	7.8301	0.2558		
	1.0080	31.2861	21.8855	9.4006	0.3105		

APPENDIX D
INSTRUMENT DETECTION LIMITS

Table D-1: Metals Detection Limits

Metal	Detection Limit (mg/L)	
	TJA Environ 60 ICP-AES	Perkin-Elmer GFAA 5100
Al	0.070	
As	0.0050	0.005
Ba	0.0070	
Ca	0.20	
Cd	0.0055	
Co	0.011	
Cr	0.017	0.001
Cu	0.014	0.005
Fe	0.045	0.005
K	0.31	
Mg	0.060	
Mn	0.011	
Na	0.15	
Ni	0.015	
Pb	0.0025	0.005
V	0.010	
Zn	0.025	

APPENDIX E
QUALITY ASSURANCE / QUALITY CONTROL

Results for the quality assurance quality control samples are presented at the end of each Appendix accordingly.

APPENDIX F
TOTAL METAL ANALYSIS RESULTS OF WOOD AND TIRE ASH

Table F-1: Total Magnesium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	83.0	5,790
	#2	73.37	4,960
	#3	90.1	5,870
Load 2	#1	76.7	5280
	#2	83.7	5300
	#3	71.2	4970
Load 3	#1	67.8	4700
	#2	68.8	4980
	#3	71.0	5170
Load 4	#1	75.6	5430
	#2	81.8	3170
	#3	72.3	3710
Load 5	#1	78.8	5350
	#2	73.9	3380
	#3	70.1	3420
Load 6	#1	66.8	4720
	#2	77.8	3880
	#3	66.8	3240
Load 7	#1	67.5	4840
	#2	64.9	3340
	#3	60.4	3110
Load 8	#1	72.9	5120
	#2	74.4	2750
	#3	67.0	3550
Mixed Load	#1	99.5	6880
	#2	100	4780
	#3	92.3	4600

Table F-2: QA/QC for Total Magnesium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<0.060	NA
Blank Spike	5.0	4.89	98
Blank Spike Dup	5.0	4.90	98
Load 1 Dup	82.2	86.3	105
Load 1 MS (10 mg/L)	92.2	93.2	110
Load 1 MSD (10 mg/L)	92.2	90.4	82

Table F-3: Total Sodium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	30.8	2,148
	#2	27.6	1,870
	#3	29.0	1,890
Load 2	#1	27.6	1,900
	#2	34.4	2,180
	#3	22.5	1,570
Load 3	#1	19.1	1,320
	#2	23.0	1,670
	#3	22.6	1,650
Load 4	#1	21.8	1,570
	#2	25.7	995
	#3	22.5	1,160
Load 5	#1	27.9	1,890
	#2	24.3	1,110
	#3	21.6	1,050
Load 6	#1	23.8	1,680
	#2	25.5	1,270
	#3	23.8	1,150
Load 7	#1	26.0	1,860
	#2	25.9	1,330
	#3	22.3	1,150
Load 8	#1	24.3	1,700
	#2	24.9	918
	#3	23.5	1,240
Mixed Load	#1	26.8	1,850
	#2	34.0	1,620
	#3	38.9	1,940

Table F-4: QA/QC for Total Sodium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<0.15	NA
Blank Spike	5.0	4.9	98
Blank Spike Dup	5.0	5.0	100
Load 1 Dup	26.4	23.2	88
Load 1 MS (10mg/L)	36.4	36.1	97
Load 1 MSD (10 mg/L)	36.4	34.8	84

Table F-5: Total Aluminum Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	114	7,950
	#2	97.1	6,570
	#3	116	7,530
Load 2	#1	112	7,760
	#2	145	9,170
	#3	89.0	6,210
Load 3	#1	82.1	5,700
	#2	102	7,360
	#3	99.0	7,210
Load 4	#1	95.2	6,840
	#2	108	4,180
	#3	98.1	5,030
Load 5	#1	103	7,000
	#2	101	4,590
	#3	88.2	4,310
Load 6	#1	102	7,230
	#2	102	5,100
	#3	102	4,960
Load 7	#1	104	7,490
	#2	107	5,530
	#3	91.6	4,730
Load 8	#1	103	7,200
	#2	103	3,790
	#3	102	5,380
Mixed Load	#1	108	7,450
	#2	139	6,610
	#3	119	5,930

Table F-6: QA/QC for Total Aluminum Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	< 0.07	NA
Blank Spike	50	49	98
Blank Spike Dup	50	45	90
Load 1 Dup	109	101	93
Load 1 MS (10 mg/L)	119	127	82
Load 1 MSD (10 mg/L)	119	128	87

Table F-7: Total Potassium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	114	7,950
	#2	97.1	6,570
	#3	116	7,530
Load 2	#1	112	7,760
	#2	145	9,170
	#3	89.0	6,210
Load 3	#1	82.1	5,700
	#2	102	7,360
	#3	99.0	7,210
Load 4	#1	95.2	6,840
	#2	108	4,180
	#3	98.1	5,030
Load 5	#1	103	7,000
	#2	101	4,590
	#3	88.2	4,310
Load 6	#1	102	7,230
	#2	102	5,100
	#3	102	4,960
Load 7	#1	104	7,490
	#2	107	5,530
	#3	91.6	4,730
Load 8	#1	103	7,200
	#2	103	3,790
	#3	102	5,380
Mixed Load	#1	108	7,450
	#2	139	6,610
	#3	119	5,930

Table F-8: QA/QC for Total Potassium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	< 0.07	NA
Blank Spike	50	49	98
Blank Spike Dup	50	45	90
Load 1 Dup	109	101	93
Load 1 MS (10 mg/L)	119	127	82
Load 1 MSD (10 mg/L)	119	128	87

Table F-9: Total Calcium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	3,570	248,970
	#2	3,090	208,990
	#3	4,010	261,050
Load 2	#1	3,270	225,790
	#2	3,720	235,170
	#3	2,970	207,110
Load 3	#1	2,520	174,650
	#2	2,540	183,830
	#3	2,710	197,090
Load 4	#1	4,010	288,010
	#2	3,840	148,910
	#3	3,210	164,800
Load 5	#1	3,230	219,120
	#2	3,310	151,380
	#3	2,940	143,720
Load 6	#1	2,420	170,950
	#2	3,510	174,720
	#3	2,420	117,320
Load 7	#1	2,430	174,370
	#2	2,140	110,400
	#3	2,100	108,300
Load 8	#1	2,640	185,420
	#2	2,690	99,230
	#3	2,570	135,870
Mixed Load	#1	4,040	279,370
	#2	4,100	195,420
	#3	3,930	195,820

Table F-10: QA/QC for Total Calcium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	100	87	87
Blank Spike Dup	100	93	93
Load 1 Dup	3,557	3,498	98
Load 1 MS (500 mg/L)	4,057	4,108	110
Load 1 MSD (500 mg/L)	4,057	3,987	86

Table F-11: Total Barium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.543	37.9
	#2	0.513	34.7
	#3	0.602	39.3
Load 2	#1	0.572	39.5
	#2	0.583	36.9
	#3	0.495	34.6
Load 3	#1	0.509	35.3
	#2	0.533	38.6
	#3	0.487	35.5
Load 4	#1	0.452	32.5
	#2	0.540	20.9
	#3	0.498	25.5
Load 5	#1	0.512	34.8
	#2	0.543	24.8
	#3	0.507	24.8
Load 6	#1	0.572	40.5
	#2	0.508	25.3
	#3	0.572	27.8
Load 7	#1	0.563	40.4
	#2	0.610	31.4
	#3	0.534	27.6
Load 8	#1	0.602	42.2
	#2	0.758	28.0
	#3	0.530	28.1
Mixed Load	#1	0.686	47.5
	#2	0.776	37.0
	#3	0.673	33.5

Table F-12: QA/QC for Total Barium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	1.0	98	98
Blank Spike Dup	1.0	1.1	110
Load 1 Dup	0.553	0.514	93
Load 1 MS (0.5 mg/L)	1.05	1.13	115
Load 1 MSD (0.5 mg/L)	1.05	1.10	107

Table F-13: Total Vanadium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.0862	6.01
	#2	0.0976	6.60
	#3	0.103	6.68
Load 2	#1	0.112	7.70
	#2	0.143	9.02
	#3	0.0984	6.87
Load 3	#1	0.145	10.1
	#2	0.108	7.80
	#3	0.0901	6.56
Load 4	#1	0.0766	5.50
	#2	0.114	4.41
	#3	0.0927	4.75
Load 5	#1	0.157	10.6
	#2	0.131	6.00
	#3	0.105	5.13
Load 6	#1	0.102	7.19
	#2	0.0872	4.35
	#3	0.102	4.93
Load 7	#1	0.144	10.3
	#2	0.189	9.74
	#3	0.111	5.71
Load 8	#1	0.163	11.4
	#2	0.151	5.58
	#3	0.127	6.74
Mixed Load	#1	0.164	11.3
	#2	0.160	7.64
	#3	0.149	7.43

Table F-14: QA/QC for Total Vanadium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	0.5	0.56	112
Blank Spike Dup	0.5	0.48	96
Load 1 Dup	0.0956	0.112	117
Load 1 MS (0.1 mg/L)	0.1956	0.204	108
Load 1 MSD (0.1 mg/L)	0.1956	0.199	103

Table F-15: Total Chromium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.548	38.2
	#2	0.565	38.2
	#3	0.679	44.2
Load 2	#1	0.634	43.8
	#2	0.677	42.8
	#3	0.626	43.7
Load 3	#1	0.648	45.0
	#2	0.672	48.6
	#3	0.555	40.4
Load 4	#1	0.540	38.8
	#2	0.661	25.6
	#3	0.585	30.0
Load 5	#1	0.691	46.9
	#2	0.542	24.8
	#3	0.517	25.3
Load 6	#1	0.608	43.0
	#2	0.497	24.8
	#3	0.608	29.5
Load 7	#1	0.869	62.4
	#2	1.01	52.0
	#3	0.50	25.6
Load 8	#1	0.81	56.8
	#2	0.71	26.2
	#3	0.62	32.8
Mixed Load	#1	0.76	52.4
	#2	0.92	43.9
	#3	0.85	42.6

Table F-16: QA/QC for Total Chromium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	1.0	0.87	87
Blank Spike Dup	1.0	1.03	103
Load 1 Dup	0.597	0.489	82
Load 1 MS (0.5 mg/L)	1.10	1.17	114
Load 1 MSD (0.5 mg/L)	1.10	1.09	98

Table F-17: Total Manganese Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	2.19	153
	#2	2.69	182
	#3	2.45	160
Load 2	#1	3.78	261
	#2	4.92	311
	#3	3.05	213
Load 3	#1	6.41	445
	#2	3.37	244
	#3	2.48	180
Load 4	#1	2.28	164
	#2	4.24	164
	#3	2.80	144
Load 5	#1	6.75	458
	#2	5.59	256
	#3	3.33	163
Load 6	#1	3.18	225
	#2	2.48	124
	#3	3.18	154
Load 7	#1	5.84	419
	#2	9.45	486
	#3	3.09	160
Load 8	#1	6.15	432
	#2	5.68	210
	#3	4.23	224
Mixed Load	#1	5.73	396
	#2	5.67	270
	#3	5.14	256

Table F-18: QA/QC for Total Manganese Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	5.0	5.0	100
Blank Spike Dup	5.0	5.1	101
Load 1 Dup	2.44	2.27	93
Load 1 MS (1.0 mg/L)	3.44	3.41	97
Load 1 MSD (1.0 mg/L)	3.44	3.29	85

Table F-19: Total Iron Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	180	12,560
	#2	273	18,490
	#3	169	10,980
Load 2	#1	463	31,950
	#2	732	46,280
	#3	328	22,880
Load 3	#1	1,010	69,910
	#2	416	30,120
	#3	205	14,920
Load 4	#1	183	13,110
	#2	518	20,070
	#3	286	14,680
Load 5	#1	1,080	73,560
	#2	655	29,950
	#3	377	18,450
Load 6	#1	309	21,840
	#2	227	11,330
	#3	309	14,990
Load 7	#1	715	51,360
	#2	1,340	68,900
	#3	385	19,850
Load 8	#1	935	65,660
	#2	721	26,630
	#3	584	30,880
Mixed Load	#1	585	40,430
	#2	683	32,530
	#3	628	31,280

Table F-20: QA/QC for Total Iron Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	100	110	110
Blank Spike Dup	100	100	100
Load 1 Dup	207	170	82
Load 1 MS (100 mg/L)	307	287	80
Load 1 MSD (100 mg/L)	307	302	95

Table F-21: Total Cobalt Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	1.95	136
	#2	1.83	124
	#3	2.02	132
Load 2	#1	1.87	129
	#2	2.12	134
	#3	1.84	129
Load 3	#1	1.60	111
	#2	1.76	127
	#3	1.75	127
Load 4	#1	1.77	127
	#2	2.16	83.7
	#3	1.84	94.6
Load 5	#1	1.92	130
	#2	1.97	90.1
	#3	1.93	94.4
Load 6	#1	1.52	107.7
	#2	2.12	106
	#3	1.52	73.9
Load 7	#1	1.61	115
	#2	1.83	94.1
	#3	1.43	73.6
Load 8	#1	1.74	122
	#2	1.87	69.0
	#3	1.70	89.9
Mixed Load	#1	2.26	156
	#2	2.46	117
	#3	2.31	115

Table F-22: QA/QC for Total Cobalt Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	1.0	98	98
Blank Spike Dup	1.0	99	99
Load 1 Dup	1.93	1.83	95
Load 1 MS (1.0 mg/L)	2.93	2.84	91
Load 1 MSD (1.0 mg/L)	2.93	2.83	90

Table F-23: Total Nickel Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.165	11.5
	#2	0.191	12.9
	#3	0.167	10.9
Load 2	#1	0.228	15.7
	#2	0.290	18.3
	#3	0.199	13.9
Load 3	#1	0.239	16.6
	#2	0.193	14.0
	#3	0.252	18.3
Load 4	#1	0.193	13.8
	#2	0.238	9.2
	#3	0.157	8.1
Load 5	#1	0.300	20.3
	#2	0.365	16.7
	#3	0.239	11.7
Load 6	#1	0.169	12.0
	#2	0.214	10.7
	#3	0.169	8.2
Load 7	#1	0.325	23.3
	#2	0.430	22.1
	#3	0.205	10.6
Load 8	#1	0.230	16.1
	#2	0.220	8.1
	#3	0.219	11.6
Mixed Load	#1	0.242	16.8
	#2	0.276	13.1
	#3	0.403	20.1

Table F-24: QA/QC for Total Nickel Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	1.0	98	98
Blank Spike Dup	1.0	97	97
Load 1 Dup	0.174	0.167	96
Load 1 MS (0.10 mg/L)	0.274	0.280	106
Load 1 MSD (0.10 mg/L)	0.274	0.287	113

Table F-25: Total Copper Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	1.90	132
	#2	1.88	127
	#3	2.01	131
Load 2	#1	2.02	139
	#2	2.94	186
	#3	1.86	130
Load 3	#1	3.53	245
	#2	2.18	158
	#3	1.74	127
Load 4	#1	1.67	120
	#2	2.46	95.3
	#3	1.89	96.8
Load 5	#1	3.36	228
	#2	2.94	134
	#3	2.00	97.6
Load 6	#1	1.90	135
	#2	1.82	90.6
	#3	1.90	92.4
Load 7	#1	2.32	166
	#2	2.60	134
	#3	2.68	138
Load 8	#1	3.15	221
	#2	2.94	108
	#3	2.73	144
Mixed Load	#1	2.83	195
	#2	3.13	149
	#3	2.83	141

Table F-26: QA/QC for Total Copper Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	5.0	4.95	99
Blank Spike Dup	5.0	5.0	100
Load 1 Dup	1.93	1.94	101
Load 1 MS (1.0 mg/L)	2.93	3.12	117
Load 1 MSD (1.0 mg/L)	2.93	2.84	91

Table F-27: Total Zinc Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	289	20,150
	#2	255	17,230
	#3	303	19,720
Load 2	#1	267	18,410
	#2	292	18,450
	#3	240	16,770
Load 3	#1	236	16,350
	#2	240	17,400
	#3	267	19,410
Load 4	#1	269	19,280
	#2	318	12,310
	#3	272	13,950
Load 5	#1	278	18,830
	#2	274	12,520
	#3	271	13,230
Load 6	#1	232	16,380
	#2	298	14,840
	#3	232	11,240
Load 7	#1	220	15,790
	#2	256	13,190
	#3	191	9,880
Load 8	#1	249	17,470
	#2	248	9,170
	#3	246	13,020
Mixed Load	#1	322	22,250
	#2	343	16,330
	#3	324	16,150

Table F-28 QA/QC for Total Zinc Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	10.0	9.87	99
Blank Spike Dup	10.0	9.56	95
Load 1 Dup	282	257	91
Load 1 MS (100 mg/L)	382	388	106
Load 1 MSD (100 mg/L)	382	391	109

Table F-29: Total Cadmium Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.0355	2.48
	#2	0.0359	2.43
	#3	0.0409	2.67
Load 2	#1	0.0348	2.40
	#2	0.0400	2.53
	#3	0.0341	2.38
Load 3	#1	0.0421	2.92
	#2	0.0385	2.79
	#3	0.0342	2.49
Load 4	#1	0.0374	2.69
	#2	0.0501	1.94
	#3	0.0351	1.80
Load 5	#1	0.0435	2.95
	#2	0.0386	1.76
	#3	0.0396	1.94
Load 6	#1	0.0308	2.18
	#2	0.0387	1.93
	#3	0.0308	1.49
Load 7	#1	0.0416	2.99
	#2	0.0487	2.51
	#3	0.0339	1.75
Load 8	#1	0.0440	3.09
	#2	0.0376	1.39
	#3	0.0349	1.85
Mixed Load	#1	0.0454	3.14
	#2	0.0532	2.54
	#3	0.0494	2.46

Table F-30: QA/QC for Total Cadmium Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	0.5	0.49	98
Blank Spike Dup	0.5	0.5	100
Load 1 Dup	0.0374	0.0299	80
Load 1 MS (0.01 mg/L)	0.0474	0.0382	82
Load 1 MSD (0.01 mg/L)	0.0474	0.0463	89

Table F-31: Total Lead Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.813	56.7
	#2	0.739	50.0
	#3	0.952	62.1
Load 2	#1	0.949	65.5
	#2	0.867	54.8
	#3	0.655	45.7
Load 3	#1	0.844	58.5
	#2	0.821	59.4
	#3	0.996	72.5
Load 4	#1	0.769	55.3
	#2	0.982	38.0
	#3	0.791	40.6
Load 5	#1	0.952	64.6
	#2	0.902	41.2
	#3	0.810	39.6
Load 6	#1	0.853	60.3
	#2	0.759	37.8
	#3	0.853	41.4
Load 7	#1	1.259	90.4
	#2	0.944	48.6
	#3	0.866	44.7
Load 8	#1	0.888	62.4
	#2	0.847	31.3
	#3	0.832	44.0
Mixed Load	#1	1.08	74.8
	#2	1.24	59.0
	#3	1.12	55.6

Table F-32: QA/QC for Total Lead Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	0.89	1.0	89
Blank Spike Dup	0.92	1.0	92
Load 1 Dup	0.793	0.835	95
Load 1 MS (0.5 mg/L)	1.30	1.33	93
Load 1 MSD (0.5 mg/L)	1.27	1.33	87

Table F-33: Total Arsenic Concentration in Wood and Tire Ash

Load Number	Sample Number	Concentration (mg/L)	Concentration (mg/kg dry mass)
Load 1	#1	0.946	47.3
	#2	0.607	30.3
	#3	0.622	31.1
Load 2	#1	0.794	39.7
	#2	0.711	35.5
	#3	0.903	45.2
Load 3	#1	0.855	42.8
	#2	0.654	32.7
	#3	0.783	39.1
Load 4	#1	0.695	34.8
	#2	0.737	36.9
	#3	0.698	34.9
Load 5	#1	0.783	39.1
	#2	0.944	47.2
	#3	0.846	42.3
Load 6	#1	0.604	30.2
	#2	0.850	42.5
	#3	0.610	30.5
Load 7	#1	0.927	46.3
	#2	0.829	41.5
	#3	0.747	37.3
Load 8	#1	0.842	42.1
	#2	0.719	35.9
	#3	0.892	44.6
Mixed Load	#1	0.894	44.7
	#2	0.858	42.9
	#3	0.948	47.4

Table F-34: QA/QC for Total Arsenic Concentration in Wood and Tire Ash

	Spike Concentration (mg/L)	Actual Concentration (mg/L)	% Recovery
Blank	0	<	NA
Blank Spike	5.0	4.8	96
Blank Spike Dup	5.0	4.9	98
Load 1 Dup	0.725	0.687	94
Load 1 MS (10 mg/L)	1.25	1.03	82
Load 1 MSD (10 mg/L)	1.25	1.15	92

APPENDIX G
BATCH LEACHING TESTS RESULTS

Table G-1: Leachable Aluminum from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	0.489	<0.007		
	#2	<0.007	<0.007		
	#3	<0.007	<0.007		
Load 2	#1	<0.007	<0.007		
	#2	<0.007	<0.007		
	#3	<0.007	<0.007		
Load 3	#1	0.436	<0.007		
	#2	<0.007	<0.007		
	#3	<0.007	<0.007		
Load 4	#1	<0.007	<0.007		
	#2	0.187	<0.007		
	#3	<0.005	<0.007		
Load 5	#1	<0.005	<0.007		
	#2	0.254	<0.007		
	#3	<0.007	<0.007		
Load 6	#1	0.489	<0.007		
	#2	<0.007	<0.007		
	#3	<0.007	<0.007		
Load 7	#1	0.436	<0.007		
	#2	<0.007	<0.007		
	#3	<0.007	<0.007		
Load 8	#1	<0.007	<0.007		
	#2	<0.007	<0.007		
	#3	<0.007	<0.007		
Mixed Load	#1	<0.007	<0.007	1.46	<0.007
	#2	<0.007	<0.007	1.21	<0.007
	#3	<0.007	<0.007	1.29	<0.007

Table G-2: QA/QC Leachable Aluminum from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	0	< 0.005	NA
Blank Spike	0.025	0.024	98 %
Blank Spike Dup	0.025	0.025	98 %
Load 2 SPLP Blank	< 0.005	< 0.005	NA
Load 2 SPLP MS	0.025	0.023	92 %
Load 2 SPLP MSD	0.025	0.022	88 %
Load 2 TCLP Blank	< 0.005	< 0.005	NA
Load 2 TCLP MS	0.025	0.020	80 %
Load 2 TCLP MSD	0.025	0.021	84 %
WET Blank	0.5	0.49	100
WET MS	1.82	1.78	87
WET MSD	1.82	1.58	98
DI Water Blank	< 0.005	< 0.005	NA
DI Water MS	0.025	0.022	88 %
DI Water MSD	0.025	0.020	80 %

Table G-3: Leachable Arsenic from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 2	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 3	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 4	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 5	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 6	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 7	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Load 8	#1	<0.005	<0.005		
	#2	<0.005	<0.005		
	#3	<0.005	<0.005		
Mixed Load	#1	<0.005	<0.005	0.100	<0.005
	#2	<0.005	<0.005	0.102	<0.005
	#3	<0.005	<0.005	0.101	<0.005

Table G-4: QA/QC Leachable Arsenic from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	0	< 0.005	NA
Blank Spike	0.025	0.024	98 %
Blank Spike Dup	0.025	0.025	98 %
Load 2 SPLP Blank	< 0.005	< 0.005	NA
Load 2 SPLP MS	0.025	0.023	92 %
Load 2 SPLP MSD	0.025	0.022	88 %
Load 2 TCLP Blank	< 0.005	< 0.005	NA
Load 2 TCLP MS	0.025	0.020	80 %
Load 2 TCLP MSD	0.025	0.021	84 %
WET Blank	0.025	0.125	99 %
WET MS	0.025	0.135	110 %
WET MSD	0.025	0.122	98 %
DI Water Blank	< 0.005	< 0.005	NA
DI Water MS	0.025	0.022	88 %
DI Water MSD	0.025	0.020	80 %

Table G-5: Leachable Magnesium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	0.231	0.593		
	#2	0.248	0.587		
	#3	0.251	0.548		
Load 2	#1	0.254	0.438		
	#2	0.0256	0.658		
	#3	0.0866	0.687		
Load 3	#1	0.0647	0.542		
	#2	0.2484	0.627		
	#3	0.268	0.601		
Load 4	#1	0.287	0.674		
	#2	0.269	0.627		
	#3	0.247	0.602		
Load 5	#1	0.287	0.538		
	#2	0.221	0.587		
	#3	0.0835	0.612		
Load 6	#1	0.248	0.584		
	#2	0.154	0.609		
	#3	0.287	0.852		
Load 7	#1	0.279	0.634		
	#2	0.539	0.754		
	#3	0.469	0.681		
Load 8	#1	0.249	0.687		
	#2	0.175	0.697		
	#3	0.0504	0.750		
Mixed Load	#1	0.913	0.393	33.2	0.306
	#2	0.183	0.392	29.9	0.200
	#3	0.178	0.343	30.4	0.219

Table G-6: QA/QC Leachable Magnesium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	< 0.060	< 0.060	NA
Blank Spike	0.5	0.49	98 %
Blank Spike Dup	0.5	0.48	96 %
Load 2 SPLP Blank	< 0.060	< 0.060	NA
Load 2 SPLP MS	0.5	0.754	66 %
Load 2 SPLP MSD	0.5	0.798	75 %
Load 2 TCLP Blank	< 0.060	< 0.060	NA
Load 2 TCLP MS	0.5	0.856	96 %
Load 2 TCLP MSD	0.5	0.870	99 %
WET Blank	< 0.060	< 0.060	NA
WET MS	5.0	36.4	106 %
WET MSD	5.0	34.8	74 %
DI Water Blank	< 0.060	< 0.060	NA
DI Water MS	0.5	0.735	48 %
DI Water MSD	0.5	0.748	101 %

Table G-7: Leachable Potassium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	301	265		
	#2	300	250		
	#3	312	255		
Load 2	#1	231	209		
	#2	235	210		
	#3	242	230		
Load 3	#1	223	260		
	#2	256	263		
	#3	230	258		
Load 4	#1	256	270		
	#2	258	275		
	#3	261	273		
Load 5	#1	320	230		
	#2	298	234		
	#3	310	240		
Load 6	#1	250	264		
	#2	252	267		
	#3	253	250		
Load 7	#1	290	275		
	#2	284	435		
	#3	277	231		
Load 8	#1	228	250		
	#2	218	248		
	#3	231	245		
Mixed Load	#1	150	159	396	157
	#2	160	169	382	169
	#3	150	172	399	171

Table G-8: QA/QC Leachable Magnesium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	< 0.130	< 0.130	NA
Blank Spike	50	0.49	98 %
Blank Spike Dup	50	0.48	96 %
Load 2 SPLP Blank	< 0.130	< 0.130	NA
Load 2 SPLP MS	50	0.754	66 %
Load 2 SPLP MSD	50	0.798	75 %
Load 2 TCLP Blank	< 0.130	< 0.130	NA
Load 2 TCLP MS	50	0.856	96 %
Load 2 TCLP MSD	50	0.870	99 %
WET Blank	< 0.130	< 0.130	NA
WET MS	50	NA	NA
WET MSD	50	NA	NA
DI Water Blank	< 0.130	< 0.130	NA
DI Water MS	50	0.735	48 %
DI Water MSD	50	0.748	101 %

Table G-9: Leachable Cobalt from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 2	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 3	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 4	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 5	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 6	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 7	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Load 8	#1	< 0.011	< 0.011		
	#2	< 0.011	< 0.011		
	#3	< 0.011	< 0.011		
Mixed Load	#1	< 0.011	< 0.011	0.680	< 0.011
	#2	< 0.011	< 0.011	0.625	< 0.011
	#3	< 0.011	< 0.011	0.656	< 0.011

Table G-10: QA/QC Leachable Cobalt from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.011	NA
Blank Spike	0.5	0.49	98 %
Blank Spike Dup	0.5	0.48	96 %
Load 2 SPLP Blank	None	< 0.011	NA
Load 2 SPLP MS	0.5	0.52	104 %
Load 2 SPLP MSD	0.5	0.45	90 %
Load 2 TCLP Blank	None	< 0.011	NA
Load 2 TCLP MS	0.5	0.48	96 %
Load 2 TCLP MSD	0.5	0.49	99 %
WET Blank	None	< 0.011	NA
WET MS	0.5	1.20	109 %
WET MSD	0.5	1.08	85 %
DI Water Blank	None	< 0.011	NA
DI Water MS	0.5	0.45	90 %
DI Water MSD	0.5	0.49	98 %

Table G-11: Leachable Cadmium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 2	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 3	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 4	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 5	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 6	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 7	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Load 8	#1	< 0.0055	< 0.0055		
	#2	< 0.0055	< 0.0055		
	#3	< 0.0055	< 0.0055		
Mixed Load	#1	< 0.0055	< 0.0055	0.680	< 0.0055
	#2	< 0.0055	< 0.0055	0.625	< 0.0055
	#3	< 0.0055	< 0.0055	0.656	< 0.0055

Table G-12: QA/QC Leachable Cadmium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	0.5	0.54	108%
Blank Spike Dup	0.5	0.58	115%
Load 2 SPLP Blank	None	< 0.0055	NA
Load 2 SPLP MS	0.5	0.55	109%
Load 2 SPLP MSD	0.5	0.59	119%
Load 2 TCLP Blank	None	< 0.0055	NA
Load 2 TCLP MS	0.5	0.52	105%
Load 2 TCLP MSD	0.5	0.42	83%
WET Blank	0.65	0.59	91%
WET MS	1.15	1.15	99%
WET MSD	1.15	1.23	116%
DI Water Blank	None	< 0.0055	NA
DI Water MS	0.5	0.50	100%
DI Water MSD	0.5	0.50	100%

Table G-13: Leachable Calcium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	1473	1355	3155	982
	#2	849	1316	2380	1116
	#3	914	1494	2715	684
Load 2	#1	1157	1388	2651	555
	#2	1108	1417	3071	588
	#3	956	1413	2549	603
Load 3	#1	1425	1281	2078	1109
	#2	986	1368	2660	1048
	#3	1277	1434	2348	935
Load 4	#1	1070	1476	2381	926
	#2	1276	1255	2546	1157
	#3	1369	1317	2734	575
Load 5	#1	1495	1299	2836	912
	#2	1387	1286	2277	657
	#3	1058	1357	2927	614
Load 6	#1	1083	1377	3086	1038
	#2	912	1423	2878	1124
	#3	1484	1459	2016	807
Load 7	#1	1356	1470	2127	1020
	#2	1253	1412	2666	706
	#3	1153	1211	3022	1054
Load 8	#1	1484	1465	2571	872
	#2	1339	1225	2764	570
	#3	851	1326	2140	1097
Mixed Load	#1	1000	1550	3000	620
	#2	1156	1450	2900	645
	#3	1058	1470	3150	630

Table G-14: QA/QC Leachable Calcium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	200	194	97%
Blank Spike Dup	200	193	96%
Load 2 SPLP Blank	1150	1288	112%
Load 2 SPLP MS	1350	1314	82%
Load 2 SPLP MSD	1350	1357	103%
Load 2 TCLP Blank	1400	1674	120%
Load 2 TCLP MS	1600	1807	113%
Load 2 TCLP MSD	1600	1870	117%
WET Blank	2970	3369	113%
WET MS	3170	3177	100%
WET MSD	3170	3325	105%
DI Water Blank	628	732	116%
DI Water MS	828	838	101%
DI Water MSD	828	760	92%

Table G-15: Leachable Zinc from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	1.07	1.18	69.2	1.76
	#2	1.83	1.15	68.2	1.20
	#3	1.19	1.09	65.3	1.78
Load 2	#1	1.21	1.44	71.5	1.98
	#2	1.00	1.04	75.8	1.15
	#3	1.89	1.33	64.8	1.18
Load 3	#1	1.42	1.73	74.3	1.59
	#2	1.99	1.80	64.2	1.61
	#3	1.33	1.16	74.9	1.43
Load 4	#1	1.11	1.40	63.3	1.17
	#2	1.13	1.89	77.5	1.84
	#3	1.93	1.38	64.2	1.08
Load 5	#1	1.50	1.22	76.3	1.39
	#2	1.39	1.54	61.6	1.71
	#3	1.84	1.01	66.2	1.06
Load 6	#1	1.15	1.99	64.8	1.45
	#2	1.70	1.45	62.8	1.73
	#3	1.08	1.46	72.8	1.58
Load 7	#1	1.55	1.04	70.7	1.67
	#2	1.14	1.95	76.8	1.69
	#3	1.83	1.13	64.7	1.66
Load 8	#1	1.76	1.47	62.7	1.54
	#2	1.51	1.47	73.0	1.16
	#3	1.14	1.25	72.7	1.19
Mixed Load	#1	1.61	2.03	72.5	1.55
	#2	1.94	1.80	65.8	1.06
	#3	1.60	1.95	69.5	1.21

Table G-16: QA/QC Leachable Zinc from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	1.1	107%
Blank Spike Dup	1.0	1.0	96%
Load 2 SPLP Blank	2.7	3.1	114%
Load 2 SPLP MS	2.7	3.9	117%
Load 2 SPLP MSD	2.7	3.7	98%
Load 2 TCLP Blank	2.7	2.8	105%
Load 2 TCLP MS	2.7	2.7	99%
Load 2 TCLP MSD	2.7	2.7	99%
WET Blank	81.0	85.6	106%
WET MS	81.0	90.0	111%
WET MSD	81.0	64.9	80%
DI Water Blank	2.1	2.1	102%
DI Water MS	2.1	1.9	90%
DI Water MSD	2.1	2.2	103%

Table G-17: Leachable Barium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (µg/L)	TCLP Concentration (µg/L)	WET Concentration (µg/L)	DI Water Concentration (µg/L)
Load 1	#1	100	248	182	151
	#2	180	242	242	126
	#3	135	108	163	127
Load 2	#1	190	187	101	240
	#2	145	130	119	230
	#3	170	141	210	175
Load 3	#1	200	245	219	153
	#2	236	120	143	142
	#3	180	120	249	162
Load 4	#1	221	114	217	133
	#2	169	171	216	203
	#3	111	210	120	126
Load 5	#1	243	144	185	132
	#2	111	116	103	227
	#3	197	240	112	157
Load 6	#1	186	117	198	101
	#2	240	143	220	166
	#3	206	153	190	207
Load 7	#1	142	128	175	219
	#2	228	147	138	124
	#3	179	111	128	138
Load 8	#1	222	218	211	192
	#2	163	169	104	191
	#3	142	190	105	200
Mixed Load	#1	176	99.8	208	176
	#2	170	92.5	157	170
	#3	309	93.5	81.6	309

Table G-18: QA/QC Leachable Barium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	0.9	92%
Blank Spike Dup	1.0	1.1	112%
Load 2 SPLP Blank	2.7	2.4	88%
Load 2 SPLP MS	2.7	3.7	100%
Load 2 SPLP MSD	2.7	3.8	115%
Load 2 TCLP Blank	2.7	2.6	95%
Load 2 TCLP MS	2.7	2.2	82%
Load 2 TCLP MSD	2.7	2.2	81%
WET Blank	81.0	75.0	93%
WET MS	81.0	86.3	106%
WET MSD	81.0	81.2	100%
DI Water Blank	2.1	2.5	118%
DI Water MS	2.1	1.9	90%
DI Water MSD	2.1	2.5	118%

Table G-19: Leachable Chromium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (µg/L)	TCLP Concentration (µg/L)	WET Concentration (µg/L)	DI Water Concentration (µg/L)
Load 1	#1	9.46	10.8	73.4	25.0
	#2	7.30	9.13	77.0	29.8
	#3	7.13	9.12	65.4	19.0
Load 2	#1	10.3	10.7	76.5	22.0
	#2	6.87	11.3	66.7	22.9
	#3	7.76	6.03	62.2	18.7
Load 3	#1	8.88	8.61	77.0	23.5
	#2	11.4	11.5	61.6	18.0
	#3	11.5	9.09	72.6	16.6
Load 4	#1	5.23	7.60	63.2	30.0
	#2	8.67	10.0	78.3	20.9
	#3	11.5	9.10	76.4	27.0
Load 5	#1	10.4	7.64	68.8	22.6
	#2	11.9	6.24	60.5	20.5
	#3	6.00	6.17	65.6	16.3
Load 6	#1	6.28	11.6	68.2	15.9
	#2	8.81	11.2	67.2	20.6
	#3	7.10	6.81	64.3	18.3
Load 7	#1	10.9	5.09	77.7	27.0
	#2	7.48	9.25	64.5	22.9
	#3	6.13	8.16	75.2	15.0
Load 8	#1	9.64	7.60	65.3	21.5
	#2	5.32	10.7	72.8	15.5
	#3	5.26	9.86	66.3	18.6
Mixed Load	#1	6.20	49.7	90.5	52.3
	#2	6.89	10.9	65.5	22.1
	#3	6.78	10.9	63.3	28.3

Table G-20: QA/QC Leachable Chromium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	1.0	101%
Blank Spike Dup	1.0	1.1	111%
Load 2 SPLP Blank	2.7	2.9	109%
Load 2 SPLP MS	2.7	3.6	91%
Load 2 SPLP MSD	2.7	3.8	109%
Load 2 TCLP Blank	2.7	2.4	87%
Load 2 TCLP MS	2.7	2.9	106%
Load 2 TCLP MSD	2.7	2.4	88%
WET Blank	81.0	95.5	118%
WET MS	81.0	86.1	106%
WET MSD	81.0	89.0	110%
DI Water Blank	2.1	1.8	86%
DI Water MS	2.1	2.2	104%
DI Water MSD	2.1	2.3	109%

Table G-21: Leachable Copper from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (µg/L)	TCLP Concentration (µg/L)	WET Concentration (µg/L)	DI Water Concentration (µg/L)
Load 1	#1	< 14	< 14	40.9	< 14
	#2	< 14	16.6	< 14	< 14
	#3	< 14	< 14	29.0	< 14
Load 2	#1	< 14	14.8	29.6	< 14
	#2	14.5	< 14	49.7	< 14
	#3	15.3	14.8	46.1	< 14
Load 3	#1	< 14	13.5	28.6	15.8
	#2	< 14	16.3	< 14	< 14
	#3	15.9	18.2	40.5	15.4
Load 4	#1	15.9	< 14	20.9	< 14
	#2	< 14	< 14	35.7	14.9
	#3	< 14	< 14	48.0	< 14
Load 5	#1	< 14	15.4	20.6	< 14
	#2	14.3	< 14	40.2	< 14
	#3	< 14	15.6	50.9	< 14
Load 6	#1	14.5	17.4	57.8	< 14
	#2	< 14	< 14	26.2	< 14
	#3	< 14	16.9	23.9	< 14
Load 7	#1	14.2	< 14	45.9	< 14
	#2	< 14	< 14	34.1	15.4
	#3	14.9	18.9	33.9	< 14
Load 8	#1	15.3	14.1	53.4	14.0
	#2	15.8	< 14	17.0	< 14
	#3	< 14	< 14	57.4	< 14
Mixed Load	#1	< 14	40.5	1,450	< 14
	#2	< 14	360	1,460	< 14
	#3	< 14	26.2	1,462	< 14

Table G-22: QA/QC Leachable Copper from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	0.9	91%
Blank Spike Dup	1.0	1.1	113%
Load 2 SPLP Blank	2.7	2.6	95%
Load 2 SPLP MS	2.7	3.7	96%
Load 2 SPLP MSD	2.7	3.8	107%
Load 2 TCLP Blank	2.7	3.0	110%
Load 2 TCLP MS	2.7	3.2	118%
Load 2 TCLP MSD	2.7	2.9	107%
WET Blank	81.0	73.7	91%
WET MS	81.0	68.8	85%
WET MSD	81.0	78.4	97%
DI Water Blank	2.1	2.4	113%
DI Water MS	2.1	2.3	108%
DI Water MSD	2.1	2.4	117%

Table G-23: Leachable Iron from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	40.9	59.1	4472	70.2
	#2	50.2	67.4	2479	43.5
	#3	47.3	80.4	1972	< 40
Load 2	#1	47.5	46.9	5770	88.6
	#2	72.7	67.8	3189	60.6
	#3	83.0	76.0	4507	70.6
Load 3	#1	68.5	80.1	4038	41.8
	#2	58.8	55.0	626	78.1
	#3	75.1	94.1	1075	43.8
Load 4	#1	78.0	45.0	6185	99.7
	#2	46.1	63.8	6400	41.8
	#3	76.8	99.4	7172	49.6
Load 5	#1	81.4	< 40	1933	60.0
	#2	96.1	52.6	5460	82.1
	#3	47.0	94.2	5643	75.7
Load 6	#1	60.6	62.3	621	99.3
	#2	68.9	70.4	1767	78.4
	#3	79.7	40.9	973	90.1
Load 7	#1	< 40	90.4	610	< 40
	#2	84.6	87.4	3092	51.9
	#3	72.1	97.7	6198	66.7
Load 8	#1	46.2	63.5	4497	44.3
	#2	45.5	77.7	4880	< 40
	#3	53.8	67.7	3345	44.0
Mixed Load	#1	102	6860	157	183
	#2	86.5	7570	56.4	66.0
	#3	89.5	7420	135.5	86.4

Table G-24: QA/QC Leachable Iron from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	0.9	92%
Blank Spike Dup	1.0	1.1	108%
Load 2 SPLP Blank	2.7	2.6	96%
Load 2 SPLP MS	2.7	3.8	106%
Load 2 SPLP MSD	2.7	3.8	107%
Load 2 TCLP Blank	2.7	2.2	83%
Load 2 TCLP MS	2.7	2.5	93%
Load 2 TCLP MSD	2.7	2.5	91%
WET Blank	81.0	94.4	117%
WET MS	81.0	65.4	81%
WET MSD	81.0	95.3	118%
DI Water Blank	2.1	2.1	101%
DI Water MS	2.1	2.2	106%
DI Water MSD	2.1	2.4	117%

Table G-25: Leachable Manganese from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	< 14	< 14	105	< 14
	#2	< 14	< 14	51.7	< 14
	#3	< 14	< 14	179	< 14
Load 2	#1	< 14	< 14	162	< 14
	#2	< 14	< 14	60.1	14.2
	#3	< 14	< 14	177	< 14
Load 3	#1	< 14	< 14	105	< 14
	#2	< 14	< 14	167	< 14
	#3	< 14	< 14	198	< 14
Load 4	#1	< 14	< 14	161	< 14
	#2	< 14	14.4	124	< 14
	#3	< 14	< 14	109	< 14
Load 5	#1	< 14	< 14	81.3	< 14
	#2	< 14	< 14	99.8	14.8
	#3	< 14	< 14	65.8	14.0
Load 6	#1	< 14	< 14	87.2	14.5
	#2	< 14	14.1	108	< 14
	#3	< 14	14.1	114	< 14
Load 7	#1	< 14	< 14	50.0	< 14
	#2	< 14	< 14	124	< 14
	#3	< 14	< 14	179	< 14
Load 8	#1	14.5	< 14	76.9	< 14
	#2	< 14	< 14	31.5	< 14
	#3	14.8	< 14	98.1	< 14
Mixed Load	#1	< 14	< 14	216	< 14
	#2	< 14	< 14	186	< 14
	#3	< 14	< 14	176	< 14

Table G-26: QA/QC Manganese from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	1.0	96%
Blank Spike Dup	1.0	1.0	99%
Load 2 SPLP Blank	2.7	2.5	93%
Load 2 SPLP MS	2.7	3.8	108%
Load 2 SPLP MSD	2.7	3.8	105%
Load 2 TCLP Blank	2.7	2.4	89%
Load 2 TCLP MS	2.7	2.6	98%
Load 2 TCLP MSD	2.7	3.2	118%
WET Blank	81.0	69.4	86%
WET MS	81.0	82.6	102%
WET MSD	81.0	67.3	83%
DI Water Blank	2.1	2.5	119%
DI Water MS	2.1	2.4	113%
DI Water MSD	2.1	2.5	118%

Table G-27: Leachable Nickel from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	18.1	< 15	46.6	< 15
	#2	< 15	19.20	33.5	< 15
	#3	< 15	< 15	27.6	< 15
Load 2	#1	15.04	14.40	30.8	< 15
	#2	15.08	< 15	43.5	14.06
	#3	15.98	19.33	42.3	< 15
Load 3	#1	16.51	15.07	24.5	< 15
	#2	15.26	< 15	31.6	< 15
	#3	< 15	< 15	25.7	16.49
Load 4	#1	18.10	16.96	23.7	15.86
	#2	17.46	< 15	28.8	< 15
	#3	< 15	< 15	55.6	< 15
Load 5	#1	< 15	17.64	26.9	< 15
	#2	19.04	17.99	45.1	< 15
	#3	16.73	< 15	27.0	16.33
Load 6	#1	< 15	< 15	48.6	< 15
	#2	16.71	< 15	37.7	18.84
	#3	15.57	< 15	53.3	14.53
Load 7	#1	19.60	< 15	33.0	< 15
	#2	< 15	18.49	56.7	19.91
	#3	< 15	< 15	41.2	< 15
Load 8	#1	< 15	< 15	43.3	18.63
	#2	19.00	19.95	26.7	< 15
	#3	< 15	< 15	47.4	16.09
Mixed Load	#1	< 15	< 15	53.0	< 15
	#2	< 15	< 15	45.4	< 15
	#3	< 15	< 15	65.8	< 15

Table G-28: QA/QC Leachable Nickel from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	1.0	96%
Blank Spike Dup	1.0	1.0	99%
Load 2 SPLP Blank	2.7	2.5	93%
Load 2 SPLP MS	2.7	3.8	108%
Load 2 SPLP MSD	2.7	3.8	105%
Load 2 TCLP Blank	2.7	2.4	89%
Load 2 TCLP MS	2.7	2.6	98%
Load 2 TCLP MSD	2.7	3.2	118%
WET Blank	81.0	69.4	86%
WET MS	81.0	82.6	102%
WET MSD	81.0	67.3	83%
DI Water Blank	2.1	2.5	119%
DI Water MS	2.1	2.4	113%
DI Water MSD	2.1	2.5	118%

Table G-29: Leachable Lead from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	47.5	33.1	55.1	50.1
	#2	74.9	75.1	125	49.1
	#3	47.4	56.5	132	70.7
Load 2	#1	62.9	70.8	180	47.9
	#2	78.9	34.2	104	40.7
	#3	84.8	38.9	157	62.2
Load 3	#1	91.2	39.5	125	89.4
	#2	84.3	55.8	91.6	58.9
	#3	50.9	62.8	173	58.2
Load 4	#1	33.9	31.8	98.4	39.6
	#2	80.1	99.7	145	63.0
	#3	95.8	92.3	64.6	41.3
Load 5	#1	51.9	31.0	93.6	47.3
	#2	61.1	61.8	48.9	68.2
	#3	62.8	42.7	125	46.4
Load 6	#1	67.4	36.2	90.4	82.4
	#2	71.6	57.8	48.2	76.1
	#3	83.1	88.5	189	98.5
Load 7	#1	48.4	59.9	185	81.0
	#2	30.9	86.9	171	35.8
	#3	69.6	90.5	120	62.8
Load 8	#1	40.5	81.5	41.8	78.9
	#2	93.5	68.2	99.7	90.0
	#3	46.5	51.3	73.4	43.9
Mixed Load	#1	46.5	55.5	62.6	< 5
	#2	46.7	60.3	184	< 5
	#3	43.4	91.1	107	< 5

Table G-30: QA/QC Leachable Lead from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	None	< 0.0055	NA
Blank Spike	1.0	1.0	96%
Blank Spike Dup	1.0	1.0	99%
Load 2 SPLP Blank	2.7	2.5	93%
Load 2 SPLP MS	2.7	3.8	108%
Load 2 SPLP MSD	2.7	3.8	105%
Load 2 TCLP Blank	2.7	2.4	89%
Load 2 TCLP MS	2.7	2.6	98%
Load 2 TCLP MSD	2.7	3.2	118%
WET Blank	81.0	69.4	86%
WET MS	81.0	82.6	102%
WET MSD	81.0	67.3	83%
DI Water Blank	2.1	2.5	119%
DI Water MS	2.1	2.4	113%
DI Water MSD	2.1	2.5	118%

Table G-31: Leachable Sodium from Wood and Tire Ash

Load Number	Sample Number	SPLP Concentration (mg/L)	TCLP Concentration (mg/L)	WET Concentration (mg/L)	DI Water Concentration (mg/L)
Load 1	#1	59.6	912		
	#2	60.1	1312		
	#3	57.6	1170		
Load 2	#1	50.1	813		
	#2	54.2	912		
	#3	46.3	866		
Load 3	#1	40.0	962		
	#2	45.4	940		
	#3	48.1	1,710		
Load 4	#1	50.1	1,100		
	#2	49.5	1,140		
	#3	50.3	1,760		
Load 5	#1	60.2	903		
	#2	58.0	929		
	#3	63.0	952		
Load 6	#1	50.6	1,010		
	#2	48.4	1,050		
	#3	55.1	1,010		
Load 7	#1	53.3	1,100		
	#2	55.9	1,730		
	#3	54.8	934		
Load 8	#1	43.2	987		
	#2	43.5	920		
	#3	44.4	967		
Mixed Load	#1	36.9	817	NA	26.1
	#2	31.4	798	NA	28.0
	#3	29.3	849	NA	28.2

Table G-32: QA/QC Leachable Sodium from Wood and Tire Ash

	Spike Concentration	Actual Concentration	% Recovery
Blank	< 0.150	< 0.150	NA
Blank Spike	0.5	0.49	98 %
Blank Spike Dup	0.5	0.48	96 %
Load 2 SPLP Blank	< 0.150	< 0.150	NA
Load 2 SPLP MS	0.5	0.754	66 %
Load 2 SPLP MSD	0.5	0.798	75 %
Load 2 TCLP Blank	< 0.150	< 0.150	NA
Load 2 TCLP MS	0.5	0.856	96 %
Load 2 TCLP MSD	0.5	0.870	99 %
WET Blank	NA	NA	NA
WET MS	NA	NA	NA
WET MSD	NA	NA	NA
DI Water Blank	< 0.150	< 0.150	NA
DI Water MS	0.5	0.735	48 %
DI Water MSD	0.5	0.748	101 %

APPENIX H
US EPA MULTIPLE EXTRACTION PROCEDURE RESTULS

Table H-1: MEP Concentrations

Metal	Sample #	Concentration (mg/L)						
		Day 1	Day 2	Day 3	Day 4	Day 5	Day 6	Day 7
Mg	1	0.252	0.891	0.789	0.296	0.575	1.44	2.01
	2	0.260	0.746	0.623	0.366	0.699	2.78	2.24
	3	0.282	0.689	0.535	0.361	0.963	1.80	2.32
Na	1	821	34.9	20.0	14.9	645	611	550
	2	832	38.1	50.4	11.3	609	628	549
	3	849	38.9	40.1	10.2	613	606	613
K	1	191	7.19	1.15	1.01	3.28	1.67	1.13
	2	180	8.10	1.10	0.928	2.71	2.14	1.18
	3	193	7.57	0.883	0.800	2.69	1.67	1.24
Ca	1	1520	836	407	138	143	104	84.4
	2	1540	808	212	111	125	116	82.9
	3	1640	836	577	105	132	102	93.9
Ba	1	0.201	0.104	0.0300	0.00925	0.0240	0.0218	0.0114
	2	0.210	0.106	0.0287	0.0124	0.0234	0.0421	0.0127
	3	0.237	0.108	0.0290	0.0126	0.0275	0.0177	0.00740
Cr	1	0.0289	0.0226	0.00760	0.0201	0.0161	0.0156	0.0132
	2	0.0302	0.0169	0.0147	0.0108	0.0122	0.0161	0.0124
	3	0.0363	0.0226	0.01035	0.0136	0.0161	0.0181	0.00845
Fe	1	0.0650	0.158	0.0545	0.0944	0.0867	0.105	0.0294
	2	0.0592	0.103	0.0774	0.0570	0.0959	0.174	0.0540
	3	0.0603	0.171	0.278	0.0740	0.165	0.058	0.0578
Cu	1	0.0168	0.0130	0.00855	< 0.014	0.00895	0.0137	< 0.014
	2	0.0189	0.0140	0.00420	< 0.014	< 0.014	0.00635	< 0.014
	3	0.0250	0.0116	< 0.014	< 0.014	0.0144	< 0.014	< 0.014
Zn	1	1.88	1.11	0.288	0.117	0.011	0.123	0.0398
	2	1.94	1.11	0.0681	0.0370	0.0742	0.218	0.0588
	3	1.97	1.01	0.0842	0.0379	0.353	0.148	0.737
Al	1	<0.07	0.249	<0.07	0.7215	3.2545	0.7605	<0.07
	2	<0.07	0.156	0.186	1.27	2.33	0.340	<0.07
	3	<0.07	0.160	0.070	1.17	1.49	0.346	<0.07
Pb	1	0.0268	0.00952	<0.005	<0.005	<0.005	<0.005	<0.005
	2	0.0553	0.0235	<0.005	<0.005	<0.005	<0.005	<0.005
	3	0.0411	0.0133	<0.005	<0.005	<0.005	<0.005	<0.005

Table H-2: MEP Final Extraction pHs

Day	Sample Number		
	1	2	3
Day 1	v	11.949	11.965
Day 2	12.254	12.259	12.263
Day 3	12.051	11.862	12.008
Day 4	11.48	11.862	12.008
Day 5	11.118	10.992	10.93
Day 6	10.549	10.426	10.441
Day 7	10.076	9.956	9.758

Table H-3: QA/QC Magnesium, Sodium, Potassium, Calcium, Barium, and Chromium MEP Extraction

Sample	Mg			Na		
	Actual	Measured	Recovery	Actual	Measured	Recovery
Blank	<0.06	<0.06		< 0.15	< 0.15	
Blank Spike	1.00	0.888	89%	10.0	8.43	84%
Blank Spikde Dup	1.00	1.11	111%	10.0	9.38	94%
Day 2 Sample #2 Dup	0.746	0.808	108%	38.1	41.0	108%
Day 2 Sample #2 spike	1.75	1.61	86%	48.1	48.2	101%
Day 2 Sample #2 spike Duplicate	1.75	2.75	100%	48.1	58.1	100%

Sample	k			Ca		
	Actual	Measured	Recovery	Actual	Measured	Recovery
Blank	< 0.13	< 0.13		< 0.2	< 0.2	
Blank Spike	5.00	5.61	112%	100	111	111%
Blank Spikde Dup	5.00	4.62	92%	100	120	120%
Day 2 Sample #2 Dup	8.10	6.62	82%	808	825	102%
Day 2 Sample #2 spike	13.1	13.0	98%	908	910	102%
Day 2 Sample #2 spike Duplicate	13.1	18.0	98%	908	1010	101%

Sample	Ba			Cr		
	Actual	Measured	Recovery	Actual	Measured	Recovery
Blank	<0.007	<0.007		<0.001	<0.001	
Blank Spike	0.100	0.11	106%	0.0200	0.0201	100%
Blank Spikde Dup	0.100	0.0808	81%	0.0200	0.0210	105%
Day 2 Sample #2 Dup	0.106	0.120	113%	0.0168	0.0182	108%
Day 2 Sample #2 spike	0.206	0.215	109%	0.0368	0.0388	110%
Day 2 Sample #2 spike Duplicate	0.206	0.299	93%	0.0368	0.0565	99%

Sample	Fe			Zn		
	Actual	Measured	Recovery	Actual	Measured	Recovery
Blank	< 0.045	< 0.045		< 0.025	< 0.025	
Blank Spike	0.100	0.111	111%	1.00	0.882	88%
Blank Spikde Dup	0.100	0.100	100%	1.00	0.830	83%
Day 2 Sample #2 Dup	0.158	0.165	104%	1.11	1.23	110%
Day 2 Sample #2 spike	0.258	0.249	91%	2.11	2.21	110%
Day 2 Sample #2 spike Duplicate	0.258	0.346	88%	2.11	2.98	87%

Table H-4: QA/QC Cadmium, Aluminum and Lead MEP Extraction

Sample	Cd			Al		
	Actual	Measured	Recovery	Actual	Measured	Recovery
Blank	< 0.045	< 0.045		< 0.025	< 0.025	
Blank Spike	0.258	0.244	94%	2.11	1.99	94%
Blank Spikde Dup	0.258	0.306	118%	2.11	1.79	85%
Day 2 Sample #2 Dup	0.00550	0.00525	95%	0.0150	0.0169	113%
Day 2 Sample #2 spike	0.264	0.996	99%	2.13	8.71	87%
Day 2 Sample #2 spike Duplicate	0.264	1.39	113%	2.13	11.1	90%
Sample	pb					
	Actual	Measured	Recovery			
Blank	< 0.005	< 0.005				
Blank Spike	0.0400	0.0371	93%			
Blank Spikde Dup	0.0400	0.0442	111%			
Day 2 Sample #2 Dup	0.235	0.257	110%			
Day 2 Sample #2 spike	0.275	0.277	106%			
Day 2 Sample #2 spike Duplicate	0.275	0.308	83%			

APPENDIX J
CONTAMINANT RELEASE AS A FUNCTION OF PH

Table J-1: Metal Concentrations at Different pHs

pH	Pb	Mg	Ba	Mn	Co	Ni
12.28	1.05	4.76	1.81	0.0732	0.355	0.162
12.30	0.85	5.66	1.54	0.152	0.194	0.237
12.40	1.44	4.01	1.91	0.0310	0.339	0.240
12.13	1.55	5.84	1.66	0.0197	0.123	0.120
12.12	1.60	4.83	1.53	0.0592	0.104	0.206
10.66	0.17	64.7	1.14	0.0817	0.0282	0.0138
12.00	1.81	8.11	2.03	0.124	0.0219	0.219
11.97	1.36	5.40	1.80	0.0197	0.0437	0.182
11.82	0.57	6.82	1.71	0.0209	0.0151	0.110
10.29	0.0661	109	0.859	0.234	0.0407	0.00562
11.73	0.484	8.80	2.05	0.0254	0.0209	0.148
7.680	0.068	0.00	1.92	20.9	7.59	1.07
6.635	0.068	4470	2.46	122	70.6	5.65
6.760	0.090	4280	2.41	89.5	67.4	5.07
7.131	0.087	4070	2.45	66.0	50.7	2.67
6.813	0.151	4130	2.40	89.9	66.3	4.59
6.599	0.119	4520	2.46	96.5	75.3	5.78
6.623	0.150		2.40	155	38.0	4.68
6.355	0.276	4600	2.46	111	80.8	6.30
5.996	0.331	4620	2.38	115	87.1	6.72
6.046	0.334	5020	2.59	131	90.4	8.06
4.196	1.10	4470	3.90	130	89.1	7.65
5.959	0.357	4990	2.57	115	90.2	6.97
6.695	0.18	4120	2.26	90.3	68.3	5.42
2.844	9.39	4640	3.82	209	99.1	12.5
3.101	7.18	5010	3.78	179	99.2	10.2
4.688	0.62	5390	3.53	151	105	8.52
3.129	6.74	5140	4.14	172	107	10.0
2.850	9.14	4970	4.00	155	104	9.75
1.850	50.4	5090	4.25	194	112	10.9
2.594	9.58	4950	4.22	265	115	11.5
2.924	10.0	4290	3.88	195	97.4	10.5
1.988	48.5	4480	3.77	169	101	10.2
1.470	55.7	4250	3.78	253	99.0	14.3
1.629	24.0		3.79	200	69.2	10.0

Table J-1: Metal Concentrations at Different pHs Continue

pH	Pb	Mg	Ba	Mn	Co	Ni
1.593	48.5	5120	4.42	212	115	12.2
1.006	59.1	4760	4.84	232	117	13.5
0.939	33.8	2630	4.03	130	63.2	11.2
1.648	22.1	0	3.98	200	89.1	10.0
1.195	51.5	0	4.07	200	81.3	10.0
1.295	66.6	4990	4.23	180	106	9.83
1.330	52.0	5390	3.97	157	111	8.49
0.934	51.8	5520	4.16	194	95.5	9.47
0.963	51.8	5130	5.11	165	109	8.23
0.972	60.7	4410	4.41	240	96.8	11.0
0.857	53.0	4960	5.09	228	105.9	10.1
0.885	34.0	5160	4.99	201	108	11.3
0.873	40.2	4860	5.05	269	106	10.0
6.112	0.300	3890	2.16	85.1	69.2	4.57
8.590	0.0248	1100	1.48	3.89	0.692	0.014
6.692	0.184	3170	2.14	69.2	51.3	5.25
7.720	0.0474	1400	1.42	20.1	18.9	0.224
7.080	0.187	3480	1.51	18.4	25.7	1.26
8.060	0.0378	1620	1.30	6.61	2.24	0.0871
9.179	0.0194	432	0.99	1.27	0.457	0.0115
11.23	0.300	19.8	1.51	0.0437	0.0186	0.0246
9.680	0.0249	263	0.75	0.766	0.123	0.00562

Table J-2: Metal Concentrations at Different pHs

pH	Cu	Zn	Al	Na	K	As
12.278	0.225	34.9	1060	1060	5240	< 0.001
12.295	0.468	33.2	855	855	3990	< 0.001
12.395	0.302	38.7	811	811	4070	< 0.001
12.133	0.231	37.3	1070	1070	5310	< 0.001
12.119	0.228	35.9	1240	1240	6320	< 0.001
10.66	0.046	37.2	966	966	4750	0.0857
11.995	0.144	36.1	1100	1100	5370	< 0.001
11.968	0.155	26.6	1120	1120	5370	< 0.001
11.816	0.296	16.6	980	980	4440	< 0.001
10.29	0.056	66.2	1260	1260	5630	0.107
11.728	0.183	15.9	1170	1170	5320	< 0.001
7.68	0.151	1600	< 0.015	< 0.015		0.122
6.635	1.431	10200	1610	1610	6690	0.156
6.76	0.992	9400	1480	1480	6350	0.102
7.131	0.263	3120	1530	1530	6650	0.141
6.813	1.46	9370	1600	1600	6970	0.141
6.599	1.99	13010	1500	1500	6410	0.152
6.623	1.17	6560	< 0.015	< 0.015		0.149
6.355	1.56	13900	1500	1500	6250	0.136
5.996	1.94	16000	1530	1530	6310	0.156
6.046	1.92	16400	1650	1650	6740	0.122
4.196	17.4	16400	1440	1440	5850	0.308
5.959	2.17	16700	1580	1580	6560	0.469
6.695	1.44	10500	1370	1370	5740	0.143
2.844	108	16700	2550	1520	6180	0.538
3.101	73.2	17100	2200	1570	6540	0.338
4.688	9.26	19000	139	1740	7230	0.221
3.129	65.8	18200	2350	1740	7100	0.981
2.85	71.3	18000	2440	1620	6650	4.22
1.85	110	19200	2980	1590	6530	5.38
2.594	117	18900	3000	1600	6430	< 0.001
2.924	97.7	16600	2440	1420	5670	5.42
1.988	91.6	17600	2580	1450	5890	1.22
1.47	126	16900	2550	1500	6090	2.54
1.629	87.1	12600	1950	< 0.015	< 0.45	4.32

Table J-2: Metal Concentrations at Different pHs Continue

pH	Cu	Zn	Al	Na	K	As
1.593	155	19560	2960	1810	7520	3.69
1.006	115	18910	2860	1600	6610	9.17
0.939	136	14290	1560	1280	6570	5.15
1.648	112	14830	2340	< 0.015	< 0.45	2.84
1.195	97.7	15380	2390	< 0.015	< 0.45	6.49
1.295	124	17560	3040	1710	7170	16.8
1.33	113	18580	3180	1680	6860	22.9
0.934	128	15950	2880	1500	6220	20.8
0.963	128	18010	3150	1630	6690	58.4
0.972	149	15760	2700	1450	5830	25.2
0.857	120	17250	3150	1740	7180	26.3
0.885	165	17700	3140	1900	8170	16.9
0.873	159	17200	2970	1760	7460	20.8
6.112	1.29	13900	22.3	1360	5510	< 0.001
8.59	0.09	578	1.86	1240	5950	0.136
6.692	0.513	4210	16.0	1190	6550	0.139
7.72	0.215	3510	3.46	1380	7390	0.143
7.08	0.201	2680	11.4	1230	8050	0.152
8.06	0.105	764	2.49	< 0.015	6040	0.194
9.179	0.0549	218	2.45	< 0.015	5220	0.194
11.23	0.0617	24	4.95	< 0.015	6650	< 0.001
9.68	0.0525	134	2.06	< 0.015	6180	0.139

APPENDIX K
CONTAMINANT CONCENTRATIONS AS A FUNCTION OF TIME

Table K-1: Sodium and Calcium Concentrations as a Function of Contact Time

Time	Na		Ca	
	Average (mg/K)	Stdev	Average (mg/K)	Stdev
1	28.9	4.25	1120	49.6
2	30.8	2.43	1150	27.5
4	33.2	2.63	1180	22.5
8	33.3	3.19	1200	42.0
12	33.1	0.700	1240	20.5
24	36.9	0.784	1250	13.8
48	38.5	2.23	1270	15.1
72	38.5	2.26	1270	15.2
96	38.5	2.00	1270	14.1
120	38.7	1.51	1270	14.1
144	38.4	0.980	1270	31.3
168	39.2	0.480	1270	32.7

Table K-2: Potassium and Barium Concentrations as a Function of Contact Time

Time	K		Ba	
	Average (mg/K)	Stdev	Average (mg/K)	Stdev
1	19.6	4.26	0.0818	0.00821
2	11.5	2.44	0.0973	0.000111
4	10.7	2.63	0.106	0.00113
8	15.6	3.19	0.137	0.00425
12	6.35	0.706	0.146	0.00172
24	4.67	0.784	0.161	0.00110
48	8.85	2.23	0.151	0.00862
72	9.00	2.26	0.154	0.00173
96	7.30	2.01	0.154	0.00187
120	4.52	1.52	0.157	0.00180
144	6.95	0.986	0.149	0.0131
168	6.86	0.484	0.154	0.0119

Table K-3: Iron and Zinc Concentrations as a Function of Contact Time

Time	Fe		Zn	
	Average (mg/K)	Stdev	Average (mg/K)	Stdev
1	0.0619	0.00349	1.86	0.0551
2	0.0751	0.0220	1.88	0.0511
4	0.0937	0.0218	1.92	0.0500
8	0.109	0.0218	1.98	0.0328
12	0.111	0.0178	1.99	0.0575
24	0.117	0.0173	1.88	0.0609
48	0.124	0.00404	1.92	0.046
72	0.121	0.00820	1.91	0.0892
96	0.114	0.0128	1.91	0.0635
120	0.119	0.00590	1.92	0.0697
144	0.119	0.0167	1.91	0.0606
168	0.113	0.017	1.91	0.0665

Table K-4: Lead and Magnesium Concentrations as a Function of Contact Time

Time	Na		Ca	
	Average (mg/K)	Stdev	Average (mg/K)	Stdev
1	0.0447	0.00476	0.196	0.00204
2	0.0477	0.00433	0.249	0.00602
4	0.0509	0.00299	0.285	0.00484
8	0.0521	0.00202	0.341	0.0221
12	0.0525	0.00293	0.379	0.0220
24	0.0531	0.00567	0.469	0.00822
48	0.0533	0.00605	0.490	0.00325
72	0.0540	0.00286	0.485	0.0118
96	0.0529	0.00283	0.492	0.0235
120	0.0532	0.00291	0.483	0.0278
144			0.485	0.0286
168			0.477	0.0247

Table K-5: Chromium Concentrations as a Function of Contact Time

Time	Na	
	Average (mg/K)	Stdev
1	28.9	4.26
2	30.8	2.44
4	33.2	2.63
8	33.3	3.19
12	33.1	0.706
24	36.9	0.784
48	38.5	2.23
72	38.5	2.26
96	38.5	2.01
120	38.7	1.52
144	38.4	0.986
168	39.2	0.484

APPENDIX L
CONTAMINANT CONCENTRATIONS AS A FUNCTION OF
LIQUID-TO-SOLID RATIO

Table L-1: Lead and Sodium Concentrations as a Function of
Liquid-to-Solid Ratio

Liquid/Solid	Pb		Na	
	Average (µg/L)	Stdev	Average (mg/L)	Stdev
3	54.3	3.95	166	8.58
5	53.4	4.25	109	7.04
10	50.2	3.15	55.9	2.51
20	47.4	4.34	37.5	14.6
50	38.1	3.28	20.2	12.8
100	26.4	2.10	8.16	2.25
200	16.1	6.77	4.47	1.60
300			2.95	0.883
400			2.02	0.630
500			1.64	1.42

Table L-2: Zinc and Iron Concentrations as a Function of
Liquid-to-Solid Ratio

Liquid/Solid	Zn		Fe	
	Average (mg/L)	Stdev	Average (µg/L)	Stdev
3	1.97	0.181	127	18.0
5	1.93	0.130	131	21.4
10	2.00	0.122	89.0	12.2
20	1.99	0.196	92.5	36.3
50	1.78	0.253	94.8	13.0
100	1.71	0.0724	88.2	9.57
200	0.968	0.0588	64.0	15.5
300	0.496	0.312	43.8	8.74
400	0.343	0.0123	48.6	13.6
500	0.268	0.172	39.3	12.7

Table L-3: Chromium and Magnesium Concentrations as a Function of Liquid-to-Solid Ratio

Liquid/Solid	Cr		Mg	
	Average ($\mu\text{g/L}$)	Stdev	Average ($\mu\text{g/L}$)	Stdev
3	6.75	0.610	354	33.6
5	6.70	0.625	320	16.2
10	4.41	0.788	309	13.6
20	4.24	0.783	214	38.4
50	4.84	4.22	176	39.1
100	4.07	0.936	141	12.9
200	2.73	0.200	122	23.3
300	3.02	0.120	113	17.0
400	1.28	0.411		
500	1.34	0.693		

Table L-4: Barium and Potassium Concentrations as a Function of Liquid-to-Solid Ratio

Liquid/Solid	Ba		K	
	Average ($\mu\text{g/L}$)	Stdev	Average (mg/L)	Stdev
3	221	19.7	819	19.0
5	208	5.96	522	8.01
10	165	15.5	277	10.6
20	148	16.2	152	9.88
50	72.0	6.63	62.4	0.259
100	45.1	2.86	32.0	1.14
200	22.0	1.46	15.7	15.7
300	15.6	3.56	11.1	11.1
400	10.3	2.72	7.78	1.92
500	7.33	3.35	3.67	6.43

Table L-5: Calcium Concentrations as a Function of Liquid-to-Solid Ratio

Liquid/Solid	Ca	
	Average (mg/L)	Stdev
3	1350	19.2
5	1330	17.6
10	1290	45.1
20	1180	66.9
50	1070	71.5
100	797	44.9
200	422	39.4
300	292	39.6
400	199	17.5
500	172	1.50

APPENDIX M
RAW LYSIMETER DATA

Table M-1: Lysimeter 1 Raw Data

Date	Volume Drained (mL)	Total Volume (mL)	L/S ratio (L/kg)	pH	Cond.	ORP	COD
12/21/2000	270	270	0.0385	11.8	40.9	-72.0	
12/25/2000	700	970	0.138	12.4	45.7	-81.0	472
12/29/2000	1790	2760	0.394	12.6	27.2	-30.0	340
1/2/2001	930	3690	0.526	12.5	17.9	-10.0	119
1/12/2001	230	3920	0.559	12.6	10.6	-47.0	
1/17/2001	750	4670	0.666	12.6	10.8	-122	136
1/22/2001	1050	5720	0.815	12.6	11.2	-80.0	94.0
1/28/2001	980	6700	0.955	12.6	10.6	-31.0	67.0
2/2/2001	1200	7900	1.13		10.2	5.00	45.0
2/8/2001	1250	9150	1.30	12.5	9.60	29.0	48.0
2/19/2001	660	9810	1.40	12.6	8.89	30.0	55.0
2/25/2001	1150	10960	1.56	12.6	9.10	8.00	97.0
3/1/2001	1000	11960	1.70	12.1	9.20	65.0	90.0
3/8/2001	1000	12960	1.85	12.2	9.30	71.0	87.0
3/15/2001							
3/26/2001	1300	14260	2.03	12.1	9.19	55.0	60.0
4/9/2001	1800	16060	2.29	12.3	9.71	34.0	54.0
4/16/2001	1100	17160	2.45	12.4	9.74	80.0	
4/23/2001	1450	18610	2.65	12.3	9.93	-18.1	48.0
5/9/2001	2440	21050	3.00	12.4	9.52	46.6	45.0
5/22/2001							
6/14/2001	3600	24650	3.51	12.4	9.50	-12.0	20.0
7/2/2001	4500	29150	4.15	12.2	8.94	-3.00	
7/29/2001	5900	35050	4.99	12.2	8.00	-6.50	23.0
8/26/2001	4600	39650	5.65	12.5	8.21	-7.71	17.0
9/27/2001	4500	44150	6.29	12.5	8.31	-24.5	15.3
10/26/2001	4300	48450	6.90	12.7	7.30	-47.0	8.20
12/11/2001	5500	53950	7.69	12.7	7.00	-120	6.40
1/14/2002	6200	60150	8.57				1.00

Table M-2: Lysimeter 1 Raw Data

Date	TDS	TOC	Cl	Sulfate	Br	Alkalinity
12/21/2000	25200	80.4	4220	2000	885	3000
12/25/2000	26500	143	5550	1950	452	4100
12/29/2000	19100	39.9	1440	1900	376	4300
1/2/2001	9260	9.93	795	1780	564	3950
1/12/2001						3500
1/17/2001	6370	6.36	500	1850	245	3500
1/22/2001	7970		608			3600
1/28/2001	6030	9.23	431	1750	65.8	3500
2/2/2001	4470		97.4	1300	33.6	3650
2/8/2001	5230		196	1380	77.6	
2/19/2001	2880	7.02	75.1	1510	28.8	2100
2/25/2001	2880		77.0	1500	29.7	1800
3/1/2001	3000		64.4	1650	27.6	2000
3/8/2001		5.71	49.8	1370	19.4	
3/15/2001						
3/26/2001		5.21				1980
4/9/2001	2320		50.2	1480	12.6	
4/16/2001		4.45				
4/23/2001		4.06				2340
5/9/2001	3540	7.51	12.4	1560	5.37	1750
5/22/2001						
6/14/2001	3400	6.32	13.7	1520	4.71	2000
7/2/2001	3370	5.09				1850
7/29/2001	3310	4.71	12.2	1490	3.75	1950
8/26/2001	3810	4.56	5.19	1650	3.75	1865
9/27/2001	2970	4.53	8.31	998	5.78	2095
10/26/2001	2560	4.28				2050
12/11/2001	2080	4.16	5.56	439	4.58	2150
1/14/2002	1730	3.96	5.01	126	4.38	2100

Table M-3: Lysimeter 1 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	AL	Pb	Zn
12/21/2000								0.006	
12/25/2000	0.644	1730	8860	1290	0.369	0.247		0.033	1.54
12/29/2000	0.453	1130	5670	1240	0.309	0.138	4.52	0.039	2.06
1/2/2001							2.01	0.033	
1/12/2001	0.400	362	1890	1190	0.251	0.149	3.74	0.047	0.988
1/17/2001	0.367	178	869	1160	0.238	0.098	1.69	0.056	0.971
1/22/2001	0.337	112	546	1260	0.242	0.203	0.215	0.055	1.15
1/28/2001	0.250	78.5	368	1150	0.240	0.053		0.054	1.23
2/2/2001	0.455	60.1	261	1250	0.235	0.143	0.179	0.047	1.23
2/8/2001	0.250	44.4	193	1180	0.228	0.104	0.233	0.058	1.30
2/19/2001	0.334	47.2	178	1150	0.230	0.087	0.181	0.055	1.09
2/25/2001	0.205	58.7	213	1180	0.237	0.132	0.128	0.047	1.56
3/1/2001	0.092	69.7	244	1190	0.239	0.031	0.150	0.046	1.61
3/8/2001							0.128	0.064	
3/15/2001									
3/26/2001	0.054	51.9	179	1170	0.219	0.042	0.157	0.059	1.39
4/9/2001	0.050	42.9	147	1120	0.208	0.104	0.117	0.074	1.55
4/16/2001	0.116	41.4	146	1150	0.211	0.087	0.410	0.068	1.58
4/23/2001	0.197	35.9	130	1140	0.203	0.098	0.506	0.052	1.60
5/9/2001	0.079	29.9	109	1110	0.179	0.185	0.368	0.072	1.63
5/22/2001									
6/14/2001	0.093	41.5	142	1160	0.151	0.084	0.308	0.080	1.72
7/2/2001	0.228	25.4	89.4	1220	0.118	0.033	0.326		1.53
7/29/2001	0.268	19.4	69.1	1240	0.112	0.049	0.328	0.065	1.55
8/26/2001	0.242	10.8	41.9	700	0.057	0.091	0.204		0.968
9/27/2001	0.448	24.7	83.5	1120	0.099	0.096	0.229	0.047	1.40
10/26/2001	0.294	16.7	62.5	1170	0.106	0.080	0.182	0.062	1.62
12/11/2001	0.303	17.6	60.3	802	0.190	1.03	0.207	0.050	0.376
1/14/2002	0.368	13.0	45.0	818	0.425	0.238	0.148	0.045	1.39

Table M-4: Lysimeter 2 Raw Data

Date	Volume Drained (mL)	Total Volume (mL)	L/S ratio (L/kg)	pH	Cond.	ORP	COD	TDS
12/21/2000	140	140	0.0200	11.7	23.0	-71.0		19700
12/25/2000	680	820	0.115	11.7	36.0	-71.0	659	28000
12/29/2000	1230	2050	0.287	12.6	37.8	-46.0	467	21000
1/2/2001	930	2980	0.418	12.6	20.4	-26.1	172	9960
1/12/2001	310	3290	0.461	12.6	11.8	-54.0		
1/17/2001	720	4010	0.562	12.7	10.8	-120	167	5300
1/22/2001	950	4960	0.695	12.8	10.5	-54.0	125	8570
1/28/2001	980	5940	0.832	12.4	11.5	-1.00	71.0	7100
2/2/2001	1100	7040	0.987	12.4	10.8	5.00	139	5600
2/8/2001	1180	8220	1.15	12.4	9.70	6.00	45.0	5530
2/19/2001	980	9200	1.29	12.4	9.41	49.0	46.0	
2/25/2001	1200	10400	1.46	12.5	9.10	33.0	188	6300
3/1/2001	1000	11400	1.60	12.1	9.20	37.0		
3/8/2001	1000	12400	1.74	12.2	9.50	9.00	61.0	4200
3/15/2001	1000	13400	1.88	12.0	9.10	20.0		
3/26/2001	1700	15100	2.12	12.3	9.19	65.0		
4/9/2001	1750	16850	2.36	12.3	9.29	65.0	56.0	3660
4/23/2001	1550	18400	2.58	12.4	9.88	5.70		3080
5/9/2001	2050	20450	2.87	12.6	9.32	59.1	42.0	3280
5/22/2001	3000	23450	3.29	12.4	9.40	-27.5	37.0	1750
6/14/2001	3400	26850	3.76	12.6	9.50	-12.0	24.0	2900
7/2/2001	4100	30950	4.34	12.4	9.28	-5.00		2950
7/29/2001	5550	36500	5.12	12.4	8.40	-11.8	17.0	3390
8/26/2001	4450	40950	5.74	12.6	8.28	-7.63	23.0	3560
9/27/2001	4700	45650	6.40	12.4	8.14	-42.7	6.00	2630
10/26/2001	4500	50150	7.03	12.9	8.20	-51.2	7.90	2060
12/11/2001	4700	54850	7.69	12.5	7.80	-102	6.20	1790
1/14/2002	6300	61150	8.57				1.0	1520

Table M-5: Lysimeter 2 Raw Data

Date	Alkalinity	TOC	Cl	Sulfate	Br
12/21/2000			6022	2010	933
12/25/2000	4900	68.0	5066	1940	817
12/29/2000	4500	105	3862	1940	659
1/2/2001	4800	40.8	3680	2000	642
1/12/2001	3900	21.7			
1/17/2001	3850	12.1	1321	2040	345
1/22/2001	3800	7.79	982	2140	297
1/28/2001	3750	5.19	637	1830	246
2/2/2001	3700	9.53	258	950	113
2/8/2001			104	1080	32.9
2/19/2001		12.3			
2/25/2001	2850	5.67	64.8	1570	27.4
3/1/2001	2550				
3/8/2001	2050		55.4	1220	24.9
3/15/2001					
3/26/2001		6.08			
4/9/2001	2200	6.10	45.1	1450	12.8
4/23/2001		5.44			
5/9/2001	2250	3.98	8.97	1540	4.34
5/22/2001	2250		11.8	1540	4.21
6/14/2001	2000	5.07			
7/2/2001	1950	5.70	15.0		
7/29/2001	1955	5.44	22.6	1570	3.5
8/26/2001	2195	5.11			
9/27/2001	2180	4.84	6.19	608	5.2
10/26/2001	2175				
12/11/2001	2200		5.41	225	4.51
1/14/2002	2100	3.68	5.47	100	4.36

Table M-6: Lysimeter 2 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									
12/29/2000									
1/2/2001									
1/12/2001	0.450	1350	6730	1370	0.290	0.180	1.91	0.621	0.017
1/17/2001									0.024
1/22/2001									0.029
1/28/2001	0.800	1260	4750	1230	0.373	0.115	1.54	0.66	0.039
2/2/2001	0.420	748	4000	835	0.238	0.132	1.27	0.469	0.041
2/8/2001	0.470	235	1030	1300	0.344	0.124	1.75	0.551	0.040
2/19/2001	0.320	158	833	1300	0.243	0.091	1.79	0.592	0.041
2/25/2001	0.320	120	546	1010	0.215	0.104	1.78	0.425	0.044
3/1/2001	0.320	90.3	285	1120	0.180	0.096	1.79	0.328	0.055
3/8/2001	0.190	129	426	1310	0.320	0.0460	2.24		0.053
3/15/2001	0.190	86.8	372	995	0.243	0.0500	1.41		0.060
3/26/2001	0.270	124	337	1180	0.207	0.0560	1.91	0.784	0.066
4/9/2001	0.290	130	316	1140	0.204	0.0840	1.74	0.483	0.051
4/23/2001	0.370	104	361	1220	0.287	0.0500	1.37	0.279	0.067
5/9/2001	0.330	85.0	320	1430	0.275	0.0640	1.63	0.203	0.046
5/22/2001	0.340	57.1	212	1550	0.249	0.0720	2.03	0.332	0.058
6/14/2001	0.280	61.7	214	1130	0.225	0.0700	1.97	0.221	0.065
7/2/2001	0.330	53.7	284	1620	0.255	0.0430	1.66	0.155	0.080
7/29/2001	0.260	65.6	155	1140	0.201	0.0580	1.89	0.194	0.049
8/26/2001	0.310	66.6	221	865	0.163	0.0750	2.08	0.146	0.061
9/27/2001	0.310	62.5	196	1210	0.108	0.0860	1.90	0.401	0.072
10/26/2001	0.350	40.8	164	825	0.125	0.0540	2.18	0.244	0.049
12/11/2001	0.220	31.2	97.7	959	0.136	0.0940	1.98	0.227	0.072
1/14/2002	0.250	25.5	96.1	952	0.146	0.109	1.47	0.103	0.055

Table M-7: Lysimeter 3 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000	550	550	0.0700	11.9	45.5	-75	339	27300
12/25/2000	580	1130	0.150	11.9	45.7	-75	475	24620
12/29/2000	1260	2390	0.320	12.7	25.4	-27	401	17740
1/2/2001	950	3340	0.450	12.6	17.3	-57	136	9020
1/12/2001	250	3590	0.490	12.6	10.0	-48		
1/17/2001	710	4300	0.580	12.7	11.5	-128	132	4833
1/22/2001	1000	5300	0.720	12.5	12.5	-84	91	6940
1/28/2001	1010	6310	0.850	12.3	10.2	-10	51	6366
2/2/2001	1000	7310	0.990	12.3	10.1	-5	49	6500
2/8/2001	980	8290	1.120	12.3	9.80	8		7633
2/19/2001	1100	9390	1.270	12.4	9.26	39	43	5966
2/25/2001	1200	10590	1.430	12.1	9.30	7	39	4766
3/1/2001	1000	11590	1.570	12.1	9.50	25	44	4733
3/8/2001	1000	12590	1.700	12.1	9.70	28	22	6066
3/15/2001	1000	13590	1.840	12.1	9.30	32		
3/26/2001	1900	15490	2.100	12.3	8.79	69		
4/9/2001	1500	16990	2.300	12.4	9.71	62	46	2920
4/23/2001	1650	18640	2.520	12.5	9.70	-31		
5/9/2001	2150	20790	2.820	12.4	9.45	41	39	1770
5/22/2001	3000	23790	3.220	12.4	9.60	-38	34	1710
6/14/2001	3900	27690	3.750	12.5	8.70	-14	31	1670
7/2/2001	4200	31890	4.320	12.4	9.15	7		1680
7/29/2001	5800	37690	5.100	12.5	7.70	-11	14	3390
8/26/2001	4900	42590	5.770	12.6	8.27	-8	10	3710
9/27/2001	4700	47290	6.400	12.5	8.30	-31	5	3100
10/26/2001	4600	51890	7.030	12.8	8.32	-45	9	2310
12/11/2001	6200	58090	7.870	12.0	8.10	-81	1.4	1730
1/14/2002	5400	63490	8.600					1750

Table M-8: Lysimeter 3 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000			6640	2180	1020
12/25/2000	116		5280	2270	842
12/29/2000	150		3190	2200	650
1/2/2001	181		1620	2780	486
1/12/2001	24.2	3450		0	
1/17/2001	23.5		931	2050	293
1/22/2001	12	3750	883	2230	286
1/28/2001	6.8	3900	738	2010	269
2/2/2001	8.4	3850	265	1930	117
2/8/2001			289	1680	122
2/19/2001		2850	43.3	1530	16.5
2/25/2001	7.4	2550	33.9	1580	15.1
3/1/2001		2050	30.8	1500	13.8
3/8/2001	7.1	2000	44.2	1520	13.8
3/15/2001					
3/26/2001	6.1				
4/9/2001	5.2	2200	24.3	1550	12.2
4/23/2001	5.4				
5/9/2001	5	2250			
5/22/2001		2250	15.7	1500	5.20
6/14/2001	6.2	2000			
7/2/2001	5.8	1950			
7/29/2001	4.6	1960	5.80	1550	3.94
8/26/2001	4.4	2200	4.49	1620	3.37
9/27/2001	4.7	2180	5.75	664	5.04
10/26/2001		2180			
12/11/2001		2150	5.07	157	4.46
1/14/2002	3.1	2150	5.22	66	4.43

Table M-9: Lysimeter 3 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000	0.744	1944	9064	1088	0.333	0.459	1.44	5.46	0.023
12/29/2000	0.606	769	4007	1069	0.393	0.413	2.30	2.00	0.044
1/2/2001	0.320	288	1936	918	0.253	0.292	1.28	3.95	0.039
1/12/2001	0.318	323	1574	1181	0.223	0.145	0.97	2.11	0.036
1/17/2001	0.365	248	1202	1473	0.319	0.161	1.36	0.249	0.039
1/22/2001	0.424	122	412	1244	0.262	0.153	1.17		0.055
1/28/2001	0.349	99.1	373	1225	0.233	0.064	1.37	0.215	0.038
2/2/2001	0.456	49.7	286	1512	0.246	0.234	1.19	0.234	0.066
2/8/2001	0.397	49.2	137	1507	0.306	0.090	1.35	0.247	0.041
2/19/2001	0.292	42.3	222	1161	0.296	0.098	1.55	0.117	0.052
2/25/2001	0.169	50.4	232	1616	0.236	0.149	1.41	0.197	0.046
3/1/2001	0.160	80.8	259	1366	0.189	0.069	2.10	0.141	0.041
3/8/2001	0.229	43.3	142	1619	0.222	0.048	1.56	0.187	0.062
3/15/2001	0.255	41.5	161	1459	0.259	0.136	1.24	0.143	0.057
3/26/2001	0.136	50.9	135	1157	0.221	0.158	1.56	0.096	0.055
4/9/2001	0.164	49.6	134	1095	0.240	0.131	2.13	0.370	0.055
4/23/2001	0.181	34.1	97	1482	0.186	0.137	1.42	0.441	0.042
5/9/2001	0.340	31.4	109	1160	0.227	0.173	2.22	0.390	0.043
5/22/2001	0.217	28.7	75.9	1530	0.235	0.072	1.61	0.427	0.057
6/14/2001	0.294	37.8	108	1523	0.186	0.122	1.69	0.376	0.049
7/2/2001	0.273	22.7	104	1134	0.104	0.047	2.15	0.271	
7/29/2001	0.354	15.5	78.9	1523	0.151	0.091	1.88	0.238	0.049
8/26/2001	0.268	23.2	55.5	861	0.103	0.121	2.02	0.263	
9/27/2001	0.143	29.7	82.9	981	0.099	0.122	1.37	0.239	0.080
10/26/2001	0.335	23.2	75.3	1314	0.119	0.098	1.77	0.253	0.084
12/11/2001	0.216	27.9	102	847	0.135	0.126	1.62	0.136	0.057
1/14/2002	0.744	1944	9064	1088	0.333	0.459	1.44	5.46	0.023

Table M-10: Lysimeter 4 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000								
12/25/2000								
12/29/2000	250	250	0.02	12.44	33.2	-40		
1/2/2001	1400	1650	0.12	12.49	35.6	-33	402	29280
1/12/2001	650	2300	0.16	12.65	36.5	-70	553	24367
1/17/2001	540	2840	0.20	12.65		-83	1145	22833
1/22/2001	1150	3990	0.28	12.63	36.4	-94	765	19267
1/28/2001	980	4970	0.35	12.65	24.3	-32	396	11667
2/2/2001	1200	6170	0.44	12.81	16.22	-1	206	9233
2/8/2001	1150	7320	0.52	12.48	12.6	-2	147	
2/19/2001	1000	8320	0.59	12.78	11.2	27	101	4040
2/25/2001	1200	9520	0.68	12.79	10.1	31	96	4660
3/1/2001	1000	10520	0.75	12.81	10.8	34	86	2540
3/8/2001								
3/15/2001	1000	11520	0.82	12.66	9.4	1	62	3340
3/26/2001	2300	13820	0.98	12.53	10.15	45		2540
4/9/2001	1400	15220	1.08	12.35	10.48	43	56	2460
4/23/2001	1550	16770	1.19	12.53	10.4	8.3		
5/9/2001	1400	18170	1.29	12.51	9.96	54.5	52	3820
5/22/2001	2900	21070	1.49	12.48	10	-24.5	48	3720
6/14/2001								
7/2/2001	2100	23170	1.64	12.18	8.94	-3	48	3616
7/29/2001	2650	25820	1.83	12.56	8.6	-12.5	37	3830
8/26/2001	2800	28620	2.03	12.58	8.96	-7.799	30	4160
9/27/2001	2875	31495	2.23	12.47	8.95	-82.2	24	3140
10/26/2001	4000	35495	2.52	12.76	7.7	-50.1	25.7	2930
12/11/2001	4000	39495	2.80	12.68		-166.6	12.6	2570
1/14/2002	5200	44695	3.17					2320

Table M-11: Lysimeter 4 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000			6640	2180	1020
12/25/2000	116		5280	2270	842.48
12/29/2000	150		3190	2200	650.18
1/2/2001	181		1620	2780	486.3
1/12/2001	24.2	3450			
1/17/2001	23.5		931	2050	293.28
1/22/2001	12	3750	883	2230	286.35
1/28/2001	6.8	3900	738	2010	269.11
2/2/2001	8.4	3850	265	1930	117.32
2/8/2001			289	1680	122.39
2/19/2001		2850	43.3	1530	16.51
2/25/2001	7.4	2550	33.9	1580	15.12
3/1/2001		2050	30.8	1500	13.78
3/8/2001	7.1	2000	44.2	1520	13.82
3/15/2001					
3/26/2001	6.1				
4/9/2001	5.2	2200	24.3	1550	12.2
4/23/2001	5.4				
5/9/2001	5	2250			
5/22/2001		2250	15.7	1500	5.2
6/14/2001	6.2	2000			
7/2/2001	5.8	1950			
7/29/2001	4.6	1955	5.80	1550	3.94
8/26/2001	4.4	2195	4.49	1620	3.37
9/27/2001	4.7	2180	5.75	664	5.04
10/26/2001		2175			
12/11/2001		2150	5.07	157	4.46
1/14/2002	3.1	2150	5.22	65.7	4.43

Table M-12: Lysimeter 4 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									
12/29/2000									
1/2/2001									
1/12/2001	0.62	1160	9400	1120	0.353	0.193	1.20	0.437	0.62
1/17/2001									
1/22/2001									
1/28/2001	0.74	1130	3040	1210	0.259	0.109	1.91	0.477	0.74
2/2/2001	0.29	617	4641	1180	0.26	0.109	1.46	0.359	0.29
2/8/2001	0.42	163	1085.9	1340	0.268	0.111	2.05	0.538	0.42
2/19/2001	0.29	116	536	1670	0.204	0.095	1.82	0.577	0.29
2/25/2001	0.23	151	539	1240	0.323	0.077	1.73	0.496	0.23
3/1/2001	0.2	96.5	345	940	0.22	0.072	1.72	0.239	0.20
3/8/2001									
3/15/2001	0.31	93.8	319	960	0.253	0.039	2.09		0.31
3/26/2001	0.34	96.0	435	1030	0.251	0.06	1.24	0.979	0.34
4/9/2001	0.38	76.6	346	1210	0.207	0.089	2.04	0.49	0.38
4/23/2001	0.29	94.0	256	1460	0.273	0.049	1.47	0.262	0.29
5/9/2001	0.34	76.2	296	1530	0.312	0.055	1.82	0.245	0.34
5/22/2001	0.26	91.1	233	1230	0.254	0.083	1.39	0.327	0.26
6/14/2001									
7/2/2001	0.33	68.4	268	1570	0.245	0.043	1.63	0.217	0.33
7/29/2001	0.31	51.5	240	1040	0.142	0.063	1.63	0.171	0.31
8/26/2001	0.28	44.3	179	1320	0.183	0.075	1.73	0.162	0.28
9/27/2001	0.21	50.3	244	1410	0.139	0.066	1.48	0.278	0.21
10/26/2001	0.38	49.5	146	880	0.168	0.064	1.46	0.275	0.38
12/11/2001	0.3	23.5	105	770	0.162	0.113	2.13	0.191	0.30
1/14/2002	0.18	43.4	125	1100	0.12	0.078	1.84	0.159	0.18

Table M-13: Lysimeter 5 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000								
12/25/2000								
12/29/2000	584	584	0.04	12.7	39.4	-67		
1/2/2001	800	1380	0.09	12.6	36.1	-37	508	29200
1/12/2001	1030	2410	0.16	12.6	36.7	-54	480	
1/17/2001	900	3310	0.22	12.7		-90	892	30100
1/22/2001	1030	4340	0.29	12.8	30.7	-83	1518	30900
1/28/2001	900	5240	0.35	12.4	24.1	-35	376	20600
2/2/2001	1250	6490	0.44	12.7	15.9	7	212	12200
2/8/2001	1100	7590	0.51	12.5	12.7	-13	144	8730
2/19/2001	1000	8590	0.58	12.7	11.1	45		7130
2/25/2001	1200	9790	0.66	12.7	10.2	33	108	6270
3/1/2001	1000	10800	0.72	12.8	9.8	45	144	9000
3/8/2001	1000	11800	0.79	12.8	10.2	52	65	7570
3/15/2001	1000	12800	0.86	12.7	10.2	50	57	
3/26/2001	2700	15500	1.04	12.3	9.9	50		
4/9/2001	1700	17200	1.15	12.4	10.2	49	55	3800
4/23/2001	1800	19000	1.27	12.4	10.4	-20.9		
5/9/2001	2000	21000	1.41	12.6	9.9	33.1	50	3520
5/22/2001	3500	24500	1.64	12.4	9.5	-50.6	41	3580
6/14/2001	3500	28000	1.88	12.6	9.5	-12	35	3320
7/2/2001	4100	32100	2.15	12.4	9.3	-5	28	2760
7/29/2001	5650	37700	2.53	12.5	8.5	-12.2	26	3490
8/26/2001	5200	42900	2.88	12.6	8.3	7.633	20	3510
9/27/2001	4900	47800	3.21	12.5	8.6	-38.4	17.9	3340
10/26/2001	4700	52500	3.53	12.8	8.2	-50.2	17.3	2920
12/11/2001	5900	58400	3.92	12.0	8.1	-80.5	11.9	2550
1/14/2002	5600	64000	4.30				8.5	2460

Table M-14: Lysimeter 5 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000					
12/25/2000					
12/29/2000	95.1		16200	3860	856
1/2/2001	136		9690	3800	956
1/12/2001	155	4750	6800	3720	433
1/17/2001	42.9	4350	5420	2590	558
1/22/2001	42.7	4300	3120	2240	320
1/28/2001	17.4	3950	688	1760	198
2/2/2001	21.0		650	1330	181
2/8/2001		4100	613	1220	167
2/19/2001	11.7		365	1220	132
2/25/2001		4000	284	1000	120
3/1/2001		4150	273	1140	116
3/8/2001			163	1300	39.4
3/15/2001	11.1				
3/26/2001	10.9	3500			
4/9/2001	9.15		56.0	990	18.6
4/23/2001	7.55	2250			
5/9/2001		2000	31.1	1710	9.70
5/22/2001	9.68	2000	33.0	1680	9.35
6/14/2001	11.2	1850	46.0	1090	8.68
7/2/2001	9.09	1850			
7/29/2001	8.97	2000	19.4	1030	6.72
8/26/2001	8.84	2135	10.4	827	6.12
9/27/2001	8.87	2140	13.3	729	8.35
10/26/2001		2155			
12/11/2001		2225	8.65	679	6.87
1/14/2002	6.39	2200	7.73	535	6.24

Table M-15: Lysimeter 5 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									0.006
12/29/2000	0.687	1435	9518	1692	0.407	0.466	2.11	5.97	0.023
1/2/2001	0.644	804	3799	1074	0.268	0.48	1.72	2.69	0.042
1/12/2001	0.478	324	1685	866	0.184	0.264	1.32	3.41	0.051
1/17/2001	0.444	362	1569	1653	0.356	0.098	0.98	1.55	0.047
1/22/2001	0.348	205	867	1400	0.272	0.181	1.07	0.203	0.057
1/28/2001	0.368	120	390	1429	0.276	0.155	1.68		0.052
2/2/2001	0.313	93.5	331	1358	0.247	0.073	1.06	0.249	0.054
2/8/2001	0.28	76.7	260	1329	0.215	0.179	1.05	0.256	0.055
2/19/2001	0.243	54.9	134	1386	0.205	0.12	2.01	0.177	0.049
2/25/2001	0.353	35	137	1235	0.289	0.08	1.64	0.165	0.069
3/1/2001	0.186	45.8	220	1388	0.24	0.11	1.62	0.16	0.046
3/8/2001	0.166	75.4	290	1417	0.223	0.05	1.74	0.157	0.058
3/15/2001	0.185	27.5	135	1067	0.177	0.061	1.46	0.181	0.052
3/26/2001	0.25	54.3	124	1201	0.237	0.14	1.76	0.125	0.04
4/9/2001	0.179	39.4	125	1013	0.193	0.11	1.75	0.118	0.07
4/23/2001	0.205	34.1	132	1134	0.277	0.102	1.91	0.672	0.072
5/9/2001	0.284	25.3	96.6	1238	0.209	0.123	2.32	0.375	0.037
5/22/2001	0.314	34.4	84.6	1675	0.169	0.15	2.14	0.319	0.071
6/14/2001	0.223	32.5	89.5	1613	0.216	0.109	1.61	0.4	0.058
7/2/2001	0.195	28.7	141	1095	0.134	0.125	2.16	0.275	0.073
7/29/2001	0.399	20	64.9	1365	0.169	0.058	2.34	0.285	
8/26/2001	0.27	17.5	56.7	1478	0.157	0.082	2.11	0.169	0.067
9/27/2001	0.224	18.4	63	923	0.095	0.09	1.64	0.264	
10/26/2001	0.186	28.2	77.6	860	0.1	0.115	1.14	0.252	0.057
12/11/2001	0.363	18.4	50.9	1402	0.126	0.109	2.11	0.167	0.09
1/14/2002	0.204	25.2	73.9	936	0.132	0.123	1.47	0.177	0.068

Table M-16: Lysimeter 6 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000								
12/25/2000								
12/29/2000	12.5	13	0.001	12.07	24	-50		
1/2/2001	1360	1373	0.09	12.47	33.3	-65	774	28620
1/12/2001	750	2123	0.15	12.64	36.5	-81	816	
1/17/2001	830	2953	0.20	12.69		-82	3300	30667
1/22/2001	970	3923	0.27	13.002	31.4	-83	2800	34860
1/28/2001	830	4753	0.33	12.558	22.9	-46	2380	20600
2/2/2001	740	5493	0.38	12.798	15.99	-10	454	11433
2/8/2001	1100	6593	0.45	12.308	12.4	-11	140	9200
2/19/2001	880	7473	0.51	12.624	10.6	39	103	7000
2/25/2001	1000	8473	0.58	12.731	10.2	47	66	6867
3/1/2001	1000	9473	0.65	12.66	10.1	30	66	4667
3/8/2001	1000	10473	0.72	12.68	10.1	24		
3/15/2001								
3/26/2001	2550	13023	0.89	12.67	9.32	56		3751
4/9/2001	800	13823	0.95	12.41	10.22	45	60	2620
4/23/2001	1400	15223	1.04	12.46	10.46	33.1		
5/9/2001	2400	17623	1.21	12.66	9.9	64	53	3520
5/22/2001	3000	20623	1.41	12.41	9.6	-34	50	3540
6/14/2001	4100	24723	1.69	12.51	10	-17		3750
7/2/2001	4600	29323	2.01	12.39	9.01	7	44	3720
7/29/2001	5800	35123	2.40	12.55	8.3	-4.9	38	3470
8/26/2001	4900	40023	2.74	12.552	8.27	7.589	26	3060
9/27/2001	100	40123	2.74	12.497	8.63	-38.4	21.8	3270
10/26/2001	7500	47623	3.26	12.924	7.84	-45.8	19	2800
12/11/2001	5000	52623	3.60	12.657	7.48	-110.5	15.2	2700
1/14/2002	6500	58223	3.98				10.5	2100

Table M-17: Lysimeter 6 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000					
12/25/2000					
12/29/2000					
1/2/2001			12918	4147	1416
1/12/2001	61.30	3450	9576	4024	1257
1/17/2001	56.25	4750	7986	3855	1066
1/22/2001	21.58		6092	3699	931
1/28/2001	43.70	4650	2014	1779	345
2/2/2001	37.80	4300	648.5	821.1	169
2/8/2001	26.54	3900	444.9	760.1	141
2/19/2001		3850	308.6	1047.5	122
2/25/2001		3950	269.2	972.6	117
3/1/2001		4300	242.5	965.0	111
3/8/2001					
3/15/2001					
3/26/2001	10.99				
4/9/2001	11.14	3750	54.7	1220.0	19.0
4/23/2001	9.84				
5/9/2001		2250			
5/22/2001		2250	29.5	1747.0	9.1
6/14/2001	10.68				
7/2/2001	10.65	1950			
7/29/2001	5.77	2000	13.1	1414.6	6.1
8/26/2001	8.06	2100	9.0	1211.9	5.7
9/27/2001	9.87	2225	9.8	1063.6	7.7
10/26/2001		2175	9.1	985.5	6.9
12/11/2001		1975	8.5	856.2	6.8
1/14/2002	7.59	2075	8.3	820.3	6.5

Table M-18: Lysimeter 6 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									
12/29/2000									
1/2/2001									
1/12/2001	0.47	1209.4	6996.5	1771.1	0.256	0.198	1.459	0.467	0.013
1/17/2001									0.017
1/22/2001									0.032
1/28/2001	0.72	844.72	3966.5	1494.8	0.360	0.111	2.063	0.542	0.043
2/2/2001	0.32	613.48	4169.7	1390.1	0.348	0.136	1.924	0.340	0.041
2/8/2001	0.37	224.15	802.9	1077.8	0.236	0.119	1.465	0.430	0.045
2/19/2001	0.35	112.82	594.9	1600.0	0.346	0.079	1.372	0.502	0.046
2/25/2001	0.25	91.35	481.44	1073.8	0.202	0.087	2.002	0.556	0.052
3/1/2001	0.30	80.91	257.30	1290.0	0.221	0.081	1.573	0.239	0.042
3/8/2001	0.22	94.33	329.49	1595.5	0.230	0.051	1.435		0.056
3/15/2001									
3/26/2001	0.33	86.64	281.16	1661.8	0.239	0.070	1.746	0.809	0.062
4/9/2001	0.35	86.17	312.51	1560.0	0.317	0.088	1.288	0.418	0.038
4/23/2001	0.31	102.50	242.69	1517.3	0.229	0.069	1.472	0.277	0.068
5/9/2001	0.39	90.23	326.48	1624.2	0.312	0.065	1.342	0.195	0.045
5/22/2001	0.27	65.97	340.15	1345.6	0.224	0.074	1.822	0.401	0.059
6/14/2001	0.32	69.80	284.60	1580.6	0.243	0.083	1.882	0.236	0.072
7/2/2001	0.28	77.96	250.45	1306.4	0.189	0.046	1.459	0.150	0.079
7/29/2001	0.23	53.37	165.00	1155.2	0.215	0.074	1.467	0.198	0.071
8/26/2001	0.30	58.48	207.66	1166.5	0.187	0.091	2.045	0.143	0.066
9/27/2001	0.26	78.16	230.56	1419.2	0.115	0.101	1.827	0.319	0.069
10/26/2001	0.23	42.68	195.36	1172.4	0.152	0.081	1.456	0.359	0.054
12/11/2001	0.21	26.60	97.96	781.8	0.130	0.072	2.184	0.205	0.060
1/14/2002	0.23	32.33	96.41	961.0	0.172	0.128	2.053	0.178	0.081

Table M-19: Lysimeter 7 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000								
12/25/2000								
12/29/2000	31	31	0.0015					
1/2/2001	450	481	0.023	12.49	34.5	-76		29400
1/12/2001	670	1151	0.055	12.64	39	-70	1340	
1/17/2001	720	1871	0.089	12.69		-82	1284	28100
1/22/2001	1100	2971	0.141	12.667	49.5	-102		39433
1/28/2001	960	3931	0.187	12.676	45.9	-61	1050	43333
2/2/2001	1200	5131	0.244	12.863	37.6	-53	866	30066
2/8/2001	800	5931	0.282	12.382	29.1	-44	1294	23400
2/19/2001	1100	7031	0.335	12.517	20.6	-4	278	
2/25/2001	1000	8031	0.382	12.668	14.5	8		12300
3/1/2001	0	8031	0.382				150	
3/8/2001	1000	9031	0.430	12.81	14.3	2		
3/15/2001	0	9031	0.430				123	5300
3/26/2001	1000	10031	0.478	12.75	12.39	38		
4/9/2001	1000	11031	0.525	12.44	11.81	24	98	4860
4/23/2001	1100	12131	0.578	12.84	12.1	-77		
5/9/2001	2000	14131	0.673	13.15	11.5	-153	105	
5/22/2001	2500	16631	0.792	12.427	11	-204	75	4200
6/14/2001	4100	20731	0.987	12.462	10.6	-8		4140
7/2/2001	3850	24581	1.17	12.375	9.41	2	34	4020
7/29/2001	5000	29581	1.41	12.672	8.5	25.4	32	3760
8/26/2001	4400	33981	1.62	12.552	8.55	7.56	21	3400
9/27/2001	4900	38881	1.85	12.471	8.67	-29.6	23.9	3220
10/26/2001	3200	42081	2.00	12.902	8	-25.1	23.2	2840
12/11/2001	4500	46581	2.22	11.98	8.01	-57.1	13.5	2670
1/14/2002	4700	51281	2.44				12.6	2600

Table M-20: Lysimeter 7 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000					
12/25/2000					
12/29/2000					
1/2/2001	29400		422.10	127.82	56.26
1/12/2001		97.85	1044.76	288.86	137.50
1/17/2001	28100	109.35	1798.34	771.95	236.30
1/22/2001	39433.3	116.75	3158.50	885.57	370.66
1/28/2001	43333.3	89.15	2718.79	974.71	381.47
2/2/2001	30066.7	64.53	3471.63	1025.00	434.52
2/8/2001	23400	62.12	3148.71	1552.99	391.38
2/19/2001			4989.58	2037.32	408.15
2/25/2001	12300		2973.16	1942.07	585.19
3/1/2001		50	4670.70	1561.77	480.82
3/8/2001		41.2	3868.52	1692.49	511.55
3/15/2001	5300		4972.74	1469.54	492.51
3/26/2001		23.34	4194.24	1676.60	410.44
4/9/2001	4860	15.31	4968.84	2682.65	542.89
4/16/2001			3875.60	1732.05	439.30
4/23/2001		15.53	4519.72	2126.54	468.01
5/9/2001		14.28	3613.16	2290.59	547.38
5/22/2001	4200		3266.83	3165.36	680.26
6/14/2001	4140	15.43	3212.60	2928.00	595.23
7/2/2001	4020	12.53	3560.71	3760.73	527.40
7/29/2001	3760	12.2	5211.16	3113.52	497.40
8/26/2001	3400	12.2	3285.70	3035.70	545.76
9/27/2001	3220	11.73	4065.77	5276.76	726.91
10/26/2001	2840		4508.89	3654.60	521.55
12/11/2001	2670		3632.49	6129.27	449.44
1/14/2002	2600	9.158	422.10	127.82	56.26

Table M-21: Lysimeter 7 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									
12/29/2000									
1/2/2001									
1/12/2001									
1/17/2001	0.739	2025	9652	1281	0.442	0.417	1.66	1.52	0.039
1/22/2001	0.486	1659	11498	1185	0.494	0.178	2.20	0.799	0.044
1/28/2001	0.534	2193	12283	925	0.381	0.232	2.32	0.680	0.030
2/2/2001	0.544	1456	7837	1156	0.408	0.186	1.48	0.598	0.049
2/8/2001	0.411	1580	8326	1258	0.278	0.146	1.58	0.301	0.055
2/19/2001	0.307	953	5266	1242	0.309	0.087	2.05	0.425	0.036
2/25/2001	0.507	594	1897	1115	0.308	0.146	2.24	0.405	0.062
3/1/2001	0.462	244	1031	1043	0.330	0.102	1.67	0.356	0.047
3/8/2001	0.434	188	754	1226	0.289	0.083	1.91	0.246	0.065
3/15/2001	0.553	189	584	1880	0.386	0.085	1.70	0.410	0.060
3/26/2001	0.325	181	543	1065	0.217	0.115	1.98	0.253	0.082
4/9/2001	0.513	162	435	1022	0.233	0.084	1.40	0.390	0.070
4/23/2001	0.324	175	465	1648	0.318	0.065	1.63	0.262	0.056
5/9/2001	0.417	133	645	1651	0.322	0.082	1.65	0.282	0.059
5/22/2001	0.323	172	450	1305	0.255	0.103	1.40	0.262	0.052
6/14/2001	0.255	116	418	1137	0.193	0.078	2.12	0.316	0.051
7/2/2001	0.384	76.1	347	1250	0.201	0.090	1.84	0.197	0.049
7/29/2001	0.429	84.6	213	1287	0.204	0.118	1.98	0.235	0.045
8/26/2001	0.342	40.5	150	1373	0.250	0.052	1.82	0.163	0.071
9/27/2001	0.228	55.3	163	1257	0.191	0.066	1.85	0.171	0.058
10/26/2001	0.347	47.2	211	952	0.215	0.035	1.90	0.349	0.057
12/11/2001	0.329	40.2	178	1052	0.148	0.080	1.88	0.158	0.058
1/14/2002	0.228	50.2	123	1288	0.197	0.072	2.15	0.134	0.056

Table M-22: Lysimeter 8 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000								
12/25/2000								
12/29/2000	14	14	0.001					
1/2/2001	550	564	0.03	12.1	29.4	-71	1258	30200
1/12/2001	790	1354	0.06	12.6	38	-64	1182	
1/17/2001	820	2174	0.10	12.7		-55	1068	28633
1/22/2001	900	3074	0.15	12.9	45.5	-83	1080	46567
1/28/2001	950	4024	0.19	12.7	46.7	-67	2894	28967
2/2/2001	620	4644	0.22	12.2	37.6	-49	788	36233
2/8/2001	1150	5794	0.28	12.4	31.5	-40		27267
2/19/2001	1200	6994	0.33	13.1	20.5	-13	285	8367
2/25/2001	1000	7994	0.38	13.2	14	19		9100
3/1/2001	1000	8994	0.43	13.1	12	1	140	8367
3/8/2001	1000	9994	0.48	13.1	11.8	4		
3/15/2001	1000	10994	0.52	13.1	12.2	24	126	
3/26/2001	2858	13852	0.66	12.6	10.44	39		
4/9/2001	1700	15552	0.74	12.4	10.52	29	98	3760
4/23/2001	1100	16652	0.79	12.9	10.5	-67		
5/9/2001	2350	19002	0.90	13.1	10.1	-101	68	3820
5/22/2001	2500	21502	1.02	12.6	10.2	-127	55	3240
6/14/2001	4100	25602	1.22	12.5	10.6	-5	45	3870
7/2/2001	4200	29802	1.42	12.4	9.32	1	38	1735
7/29/2001	5200	35002	1.67	12.7	8.2	30.3	36	2850
8/26/2001	4100	39102	1.86	12.6	8.27	7.57	34	3280
9/27/2001	4900	44002	2.10	12.4	8.58	-29.9	30	2980
10/26/2001	4500	48502	2.31	12.9	8.24	-25.6	20	2860
12/11/2001	5900	54402	2.59	12.4	8.19	-5.8	17	2620
1/14/2002	5800	60202	2.87				11	2610

Table M-23: Lysimeter 8 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000					
12/25/2000					
12/29/2000					
1/2/2001		4250	1474	2368	362
1/12/2001			16853	5464	2168
1/17/2001	118	5000	15627	5871	2094
1/22/2001	126		7190	3389	919
1/28/2001	125	5750	4549	2843	614
2/2/2001	129	5400	4019	2558	549
2/8/2001	118	5100	3496	2984	496
2/19/2001		4000	3134	2675	448
2/25/2001	51.97	4100	493	1925	147
3/1/2001	40.00	4550	364	1981	151
3/8/2001	35.00		3093	1472	491
3/15/2001	24.00	3625	4937	2131	620
3/26/2001	15.32		4704	2456	669
4/9/2001	13.29	2375	3724	2091	428
4/16/2001			3129	1822	583
4/23/2001	8.99	2250	3093	1472	491
5/9/2001	9.32	2000	117	1883	21.6
5/22/2001		2000	50.2	1781	13.1
6/14/2001			92.8	1025	15.9
7/2/2001	11.62	1950			
7/29/2001	12.54	2250	28.7	1109	8.89
8/26/2001	10.70	2225			
9/27/2001	9.94	2020	21.3	1260	11.34
10/26/2001					
12/11/2001		2175	15.4	1115	9.73
1/14/2002	9.29	1750	10.3	640	7.46

Table M-24: Lysimeter 8 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									
12/29/2000									
1/2/2001									
1/12/2001	0.647	1669	13897	1416	0.550	0.416	1.50	2.05	0.036
1/17/2001	0.646	2436	13206	1070	0.413	0.217	2.17	0.790	0.031
1/22/2001	0.482	1842	9424	1453	0.364	0.181	2.33	0.827	0.043
1/28/2001	0.515	2090	7450	1138	0.357	0.114	2.02	0.668	0.041
2/2/2001	0.392	1218	6410	1100	0.395	0.109	1.63	0.235	0.045
2/8/2001	0.419	791	3990	1277	0.302	0.083	1.57	0.270	0.057
2/19/2001	0.418	396	2598	808	0.323	0.096	1.44	0.357	0.046
2/25/2001	0.542	200	1375	1597	0.372	0.095	1.67	0.292	0.055
3/1/2001	0.491	257	905	1572	0.251	0.107	1.95	0.313	0.052
3/8/2001	0.821	183	533	1488	0.311	0.128	2.18	0.524	0.048
3/15/2001	0.380	186	870	1342	0.220	0.119	1.38	0.314	0.059
3/26/2001	0.591	138	540	1405	0.268	0.084	2.14	0.353	0.048
4/9/2001	0.467	179	651	1070	0.204	0.103	1.42	0.279	0.066
4/23/2001	0.352	131	462	1196	0.209	0.068	1.54	0.291	0.063
5/9/2001	0.393	161	533	1348	0.314	0.094	1.96	0.204	0.065
5/22/2001	0.401	104	361	1025	0.246	0.096	1.77	0.448	0.049
6/14/2001	0.279	73	269	1022	0.204	0.059	1.88	0.213	0.056
7/2/2001	0.344	83.9	214	1053	0.249	0.120	1.39	0.238	0.049
7/29/2001	0.357	56.7	177	1467	0.211	0.073	2.42	0.126	0.047
8/26/2001	0.210	47.2	234	1073	0.231	0.067	2.19	0.198	0.055
9/27/2001	0.358	72.3	194	1152	0.209	0.041	1.88	0.274	0.063
10/26/2001	0.234	41.6	207	1321	0.227	0.099	1.95	0.219	0.080
12/11/2001	0.201	54.5	141	1110	0.141	0.057	1.51	0.196	0.044
1/14/2002	0.197	16.1	64	763	0.126	0.109	1.32	0.442	0.049

Table M-25: Lysimeter 9 Raw Data

Date	Drained Volume	Total Volume	L/S ratio	pH	Cond.	ORP	COD	TDS
12/21/2000								
12/25/2000								
12/29/2000								
1/2/2001	550	550	0.026	12.1	28.6	-60	774	31120
1/12/2001	460	1010	0.048	12.5	38	-68	816	
1/17/2001	800	1810	0.086	12.7	44	44	3300	26733
1/22/2001	850	2660	0.13	12.9	46.4	-82	2800	44167
1/28/2001	950	3610	0.17	12.7	47.7	-69	2380	48933
2/2/2001	690	4300	0.20	12.8	42.9	-64	454	38767
2/8/2001	1130	5430	0.26	12.4	35.1	-48	140	30633
2/19/2001	800	6230	0.30	13.1	23.4	-21		
2/25/2001	1050	7280	0.35	13.1	16.7	-13	103	10300
3/1/2001	1000	8280	0.39	13.1	13.2	33	66	9633.3
3/8/2001	1000	9280	0.44	13.1	12.8	16	66	
3/15/2001								
3/26/2001	1100	10380	0.49	12.8	10.88	34		8400
4/9/2001								
4/16/2001	1800	12180	0.58	13.0	11.3	-123	60	
4/23/2001	2100	14280	0.68	13.1	10.8	-81	53	4180
5/9/2001	3000	17280	0.82	12.3	10.5	-156	50	3920
5/22/2001	3500	20780	0.99	12.5	10.6	-5		4040
6/14/2001	4000	24780	1.2	12.4	9.46	5		3700
7/2/2001	5400	30180	1.4	12.7	8.6	18.3	44	3550
7/29/2001	5400	35580	1.7	12.6	8.5	12.3		3530
8/26/2001	4300	39880	1.9	12.6	8.48	7.53	39	3290
9/27/2001	5600	45480	2.2	12.5	8.78	-28.7		2980
10/26/2001	4000	49480	2.4	12.9	8.1	-25.3	23.2	2850
12/11/2001	6200	55680	2.7	12.4	8	-8.9	17.9	2950
1/14/2002	5800	61480	2.9					

Table M-26: Lysimeter 9 Raw Data

Date	TOC	Alkalinity	Cl	Sulfate	Br
12/21/2000					
12/25/2000					
12/29/2000					
1/2/2001			17500	4580	2330
1/12/2001		3450			
1/17/2001	123	4750	13500	4550	1770
1/22/2001	119		15600	6130	2010
1/28/2001	132	4650	11000	4980	1460
2/2/2001	73.7	4300			
2/8/2001	84.7	3900	3280	3280	587
2/19/2001					
2/25/2001	55.8	3850	1970	1910	332
3/1/2001		3950	541	1240	154
3/8/2001		4300	461	1250	147
3/15/2001					
3/26/2001	17.2	3750			
4/9/2001					
4/16/2001	15.6	2850	250	1860	50.4
4/23/2001	15.3	2250			
5/9/2001	13.6	2250	150	1810	30.3
5/22/2001		2010	83.8	1840	17.5
6/14/2001	14.2	2000	50.5	1520	15.2
7/2/2001	14.6	1950			
7/29/2001		2250	27.5	1230	8.72
8/26/2001	9.31	2225	20.7	1450	9.4
9/27/2001	9.73	2135	19.5	1000	9.98
10/26/2001					
12/11/2001		2175	18.4	730	10.4
1/14/2002	8.51	2200	12.8	826	8.83

Table M-27: Lysimeter 9 Raw Data

Date	Mg	Na	K	Ca	Ba	Fe	Zn	Al	Pb
12/21/2000									
12/25/2000									
12/29/2000									
1/2/2001									
1/12/2001	0.526	2355	12197	1512	0.522	0.441	2.37	2.33	0.033
1/17/2001	0.529	2210	13632	1702	0.556	0.234	1.96	0.685	0.041
1/22/2001	0.561	1738	10603	1075	0.401	0.158	2.14	0.770	0.045
1/28/2001	0.633	1836	8064	1126	0.380	0.111	1.98	0.432	0.042
2/2/2001	0.343	1226	8327	865	0.310	0.132	1.94	0.389	0.041
2/8/2001	0.308	976	3912	1184	0.382	0.080	1.41	0.398	0.053
2/19/2001	0.612	507	2697	979	0.313	0.142	2.17	0.396	0.063
2/25/2001	0.462	268	1057	1269	0.251	0.106	1.81	0.434	0.043
3/1/2001	0.407	195	1004	966	0.284	0.124	1.57	0.246	0.057
3/8/2001	0.678	140	633	1455	0.328	0.097	2.26	0.488	0.058
3/15/2001	0.416	206	890	943	0.347	0.108	1.29	0.385	0.070
3/26/2001	0.446	144	539	1530	0.319	0.081	1.26	0.321	0.074
4/9/2001	0.428	131	538	1454	0.330	0.065	1.79	0.316	0.053
4/23/2001	0.324	195	395	1003	0.290	0.083	1.46	0.254	0.067
5/9/2001	0.295	173	477	1472	0.210	0.083	1.67	0.242	0.067
5/22/2001	0.335	120	385	1224	0.296	0.079	1.64	0.430	0.054
6/14/2001	0.289	97.1	246	1163	0.300	0.078	1.78	0.180	0.066
7/2/2001	0.412	80.4	317	1471	0.185	0.099	2.13	0.350	0.064
7/29/2001	0.282	46.7	186	977	0.195	0.085	2.53	0.151	0.048
8/26/2001	0.278	61.7	249	1107	0.210	0.058	1.50	0.236	0.048
9/27/2001	0.298	64.6	163	1248	0.178	0.039	1.85	0.336	0.057
10/26/2001	0.270	65.5	143	907	0.175	0.072	1.78	0.213	0.056
12/11/2001	0.190	43.6	128	1021	0.205	0.067	1.66	0.212	0.051
1/14/2002	0.263	15.6	71	693	0.185	0.138	1.78	0.437	0.071

APPENDIX O
WOOD AND TIRE ASH MASS ADDED TO EACH LYSIMETER

Table O-1: Contribution of Ash Loads to 0.3m Lysimeters

0.3 m (1 ft) ash										
#	Height	Lysimeter #1			Lysimeter #2			Lysimeter #3		
		Initial Weight	Final Weight	Mass Added	Initial Weight	Final Weight	Mass Added	Initial Weight	Final Weight	Mass Added
1	1.5	1100	315	785	1321	237	1084	1467	578	890
2	1.5	1176	462	714	1407	914	493	1075	211	864
3	1.5	3206	2386	821	2386	1565	820	1271	330	941
4	1.5	3087	1963	1124	1963	1007	956	1007	374	633
5	1.5	1261	398	862	1816	930	886	821	492	1259
6	1.5	2847	2071	776	2071	681	1390	681	0	681
7	1.5	1505	157	1349	2513	1469	1045	1469	557	911
8	1.5	1268	679	589	679	191	489	1550	345	1205
Tot	12			7019			7163			7384
V(ft ³)	0.2			15.4			15.7			16.2
Density (lb/ft ³)				78.6			80.4			82.7

Table O-2: Contribution of Ash Loads to 0.6m Lysimeters

0.6 m (2 ft) ash										
#	Height	Lysimeter #1			Lysimeter #2			Lysimeter #3		
		Initial Weight	Final Weight	Mass Added	Initial Weight	Final Weight	Mass Added	Initial Weight	Final Weight	Mass Added
1	1.5	3211	1064	2146	3509	1225	2284	2685	492	2193
2	1.5	2818	283	2535	3188	415	2773	3729	1096	2633
3	1.5	3427	838	2589	3159	634	2525	3095	373	2723
4	1.5	2966	490	2477	3598	893	2705	3290	986	2303
5	1.5	3580	497	3083	3275	1034	2241	3172	523	2649
6	1.5	3492	1551	1941	2621		2621	3593	811	2782
7	1.5	4012	1186	2826	4703	1611	3092	3055	347	2709
8	1.5	3849	1033	2816	3269	559	2710	3586	874	2712
Tot	12			20413			20951			20704
V(ft ³)	0.20			45			46			46
Density (lb/ft ³)				79.0			83.5			81.9

Table O-3: Contribution of Ash Loads to 0.9m Lysimeters

0.9 m (3 ft) ash										
Height	Lysimeter #1			Lysimeter #2			Lysimeter #3			
	Initial Weight	Final Weight	Mass Added	Initial Weight	Final Weight	Mass Added	Initial Weight	Final Weight	Mass Added	
1	1.5	3211	1064	2146	3509	1225	2284	2685	492	2193
2	1.5	2818	283	2535	3188	415	2773	3729	1096	2633
3	1.5	3427	838	2589	3159	634	2525	3095	373	2723
4	1.5	2966	490	2477	3598	893	2705	3290	986	2303
5	1.5	3580	497	3083	3275	1034	2241	3172	523	2649
6	1.5	3492	1551	1941	2621		2621	3593	811	2782
7	1.5	4012	1186	2826	4703	1611	3092	3055	347	2709
8	1.5	3849	1033	2816	3269	559	2710	3586	874	2712
Tota	12		20413				20951			20704
V(ft ³)	0.1963		45				46			45
Density (lb/ft ³)		76.2			78.2			77.3		

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BIOGRAPHICAL SKETCH

Thabet Tolaymat was born on December 1, 1971, in Damascus, Syria. He is the son of Farouk and Shadia Tolaymat. His father owns and operates an electrical engineering firm and his mother is a homemaker.

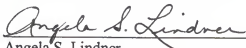
In 1984, Thabet moved to Jacksonville, Florida, with his family and attended Episcopal High School of Jacksonville. In 1991, Thabet enrolled at the University of Florida. He received a B.S. in Environmental Engineering in May 1995 and M.E in environmental engineering in 1997. He passed his Ph.D. candidacy exam in 2001. After graduation, Thabet will be joining the Environmental Protection Agency Office of Research and Development as a post-doctoral researcher.

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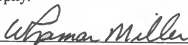
Timothy G. Townsend, Chairman
Associate Professor of Environmental
Engineering Sciences

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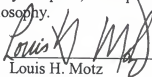
Angela S. Lindner
Assistant Professor of Environmental
Engineering Sciences

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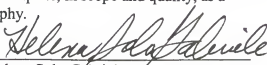
W. Lamar Miller
Professor of Environmental Engineering
Sciences

I certify that I have read this study and that in my opinion it conforms to acceptable standards of scholarly presentation and is fully adequate, in scope and quality, as a dissertation for the degree of Doctor of Philosophy.



Louis H. Motz
Associate Professor of Civil and Coastal
Engineering

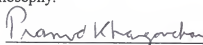
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Helena Solo-Gabriele
Associate Professor of Civil, Architecture
and Environmental Engineering
University of Miami

This dissertation was submitted to the Graduate Faculty of the College of Engineering and to the Graduate School and was accepted as partial fulfillment of the requirements for the degree of Doctor of Philosophy.

December 2003



Pramod P. Khargonekar
Dean, College of Engineering

Winfred M. Phillips
Dean, Graduate School